RELATIONSHIP BETWEEN WATER QUALITY PARAMETERS (NUTRIENTS, SESTON, CHLOROPHYLL A), HYDRODYNAMICS AND OYSTER GROWTH IN THREE MAJOR PACIFIC OYSTER (*CRASSOSTREA GIGAS*) GROWING AREAS IN SOUTHERN TASMANIA (AUSTRALIA).

By

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Submitted in fulfilment of the requirements for the degree of

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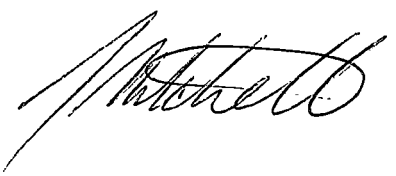

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June 2001

1. ABSTRACT

An assessment was made of three Pacific oyster (*Crassostrea gigas*) growing areas in southern Tasmania (Pitt Water, Pipeclay Lagoon and Little Swanport) with respect to water quality parameters, oyster growth and hydrodynamic characteristics. This was done in an order to explain differences in reported oyster growth rates and hence address the issue of shellfish productivity in each area. Water samples were collected monthly for 13 months from several sites along the length of each area from a marine site to the upper reaches of the estuary, or coastal embayment. These were analysed for chlorophyll a, nutrients (NOX, PO_4-P and SiO_4-Si), and seston quality and quantity (i.e. total particulate matter (TPM) and particulate organic and inorganic matter (POM & PIM)). Temperature, salinity and secchi disk depths were also recorded. Oyster growth and condition were assessed from studies conducted over three consecutive periods at two sites within each area. Hydrodynamic characteristics were calculated from tide gauge data obtained. Additionally, a biodeposition study was conducted at one area during two seasons to determine rates of deposition and composition of biodeposits.

Seston quantity was similar among areas, but seston quality, as expressed as %POM, showed variation attributed to the characteristics of each area. Chlorophyll a concentrations were generally low in each area, ranging from 0.2 to 4.0 $\mu g\ L^{-1}$. Interestingly, chlorophyll a levels measured were high in winter to early spring months within each area. Higher levels of chlorophyll a were measured following periods of flooding and freshwater inflows, particularly in two of the study areas. Considerable variation among areas was shown in oyster growth, with respect to shell length, width, depth and live weight of oysters. Differences in growth are largely attributed to the water quality and hydrodynamic characteristics noted within each of the areas. Mean biodeposition rates varied from 39.6 g DW $m^{-2}\ d^{-1}$ in winter to 180.5 g DW $m^{-2}\ d^{-1}$ in summer. The average organic content of biodeposits (approximately 19.2% POM) was similar in summer and winter. The organic matter content of sediments under oyster baskets was low (< 2.6 %), and it was concluded that biodeposits were being transported and deposited elsewhere.

The overall findings from the study indicated that growth rates and productivity of each area were largely influenced by the supply and availability of food. It appeared that

stocking density and spatial arrangement of leases provided the greater limitation on growth rate in Pitt Water and Pipeclay Lagoon. Little Swanport was characterised as having the better growth rates and conditions for growth. Food quality, as measured by chlorophyll a and %POM in particular, was higher than the other two sites, and flow rates indicated that a greater quantity of food was reaching a larger proportion of the cultured population. The marine nature of Pipeclay Lagoon suggested that the main source of food supply to the cultured oyster population is of marine origin. However, flow rates and transport of this material over the culture area is insufficient to provide faster growth rates. Stocking density of oysters, and spatial arrangement of the culture area, is most likely responsible for limitation on available food supply to the majority of the population. Sufficient food is available for maintaining metabolic processes, but is insufficient to enable greater storage and hence growth rates. Similar processes appeared to be occurring in Upper Pitt Water, though it seems the greater fraction of food is sourced from within the estuary, rather than being of marine origin. Sampling during this study was fortunate to coincide with infrequent events of heavy and prolonged rainfalls in the latter part of the year, resulting in flooding of this estuary. The beneficial effects of this were elevated nutrient, chlorophyll a, seston levels and greater increase in oyster dry meat weights, confirming the concerns raised by the oyster farmers with respect to the negative effects of the Craigbourne Dam.

Shellfish production estimates as used overseas were found to be not applicable to Tasmanian conditions. Differences in culture environments between overseas oyster growing areas and those found within Tasmania are discussed.

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1. Introduction

1.1 General overview of studies on environmental aspects of shellfish culture with particular reference to oysters

Shellfish are filter feeders dependent on phytoplankton and particulate matter from the surrounding water in which they grow. While farmers can control some aspects of shellfish growth by, for example, altering the height at which they are grown and hence immersion time, they are still ultimately reliant on food available in the water column. Other factors which influence growth are temperature, salinity, turbidity, water flow, stocking density and method of culture, that is intertidal or sub-tidal (Brown and Hartwick, 1988a; Roland and Brown, 1990; Hickman et al., 1991; Maguire et al., 1994; Grant, 1996; Soniat et al., 1998; Heasman et al., 1998; Toro et al., 1999).

Shellfish, when population numbers are sufficient, can have a significant impact on the primary productivity of an estuarine system due to their ability to filter a large volume of the water in a relatively short time (Powell et al., 1992), considerably depleting the seston of the overlying waters (Cloern, 1982; Frechette and Bourget, 1985a; McLusky and McIntyre, 1988; Carver and Mallet, 1990; Dame et al., 1991). The term 'seston' refers to all particulate material in the water column which includes living organisms (phytoplankton, zooplankton) and detritus (both inorganic and organic) (Parsons et al., 1977). Measurements of particle removal, or filtration, have been based on *in situ* field studies or laboratory experiments. These studies have assessed many factors, such as rate of uptake (or depletion), particle types and sizes filtered and ingested, assimilation and biodeposition.

The primary production and levels of nutrients in an area are important factors influencing "standing stock" of phytoplankton. Transport of seston is also important as well as the quality and quantity in the water column (Berg and Newell, 1986; Aksnes et al., 1989; Fegley et al., 1992). Thus, knowledge of current flow rates and directions, fresh water inputs and tidal exchanges are important factors for assessing the level of food availability. Assessment of primary production is based on the collection of data such as chlorophyll a, particulate matter, particulate carbon/nitrogen and nutrients (Joint and Pomroy, 1981; Rodhouse et al., 1984; Frechette and Bourget, 1985a; Berg and Newell, 1986; Grizzle and Lutz, 1989; Carver and Mallet, 1990; Ball et al., 1997).

Many of these investigations involve samples from the water column, however, the benthos plays an important role in nutrient cycling (Pomroy et al., 1983; Nowicke and Nixon, 1985; McLusky and McIntyre, 1988; Rizzo, 1990).

Shellfish, in filtering particles from the water column, deposit material (faeces and pseudofaeces) which becomes available to the benthos. The rate of biodeposition and type (i.e. amount of faeces and pseudofaeces) is dependent on a number of factors, primarily the amount of seston and quality in the water column (Sornin et al., 1983; Iglesias et al., 1998) and water temperature (Haven and Morales-Alamo, 1966). It has been found in Marennes-Oléron Bay (SW France) that bivalve biofiltration, biodeposition and subsequent mineralization may stimulate phytoplankton turnover, enhancing the carrying capacity of the ecosystem (Smaal and Zurburg, 1997).

Shellfish culture is carried out in estuaries or near-coastal waters utilising either intertidal or sub-tidal culture techniques. With intertidal culture, shellfish are frequently exposed out of the water for periods of time due to tidal variations. The duration of this exposure period can be regulated by the height of the structures upon which the animals are held. However, it is recommended that structures be not too close to the substrate where problems with predation or invasion by 'mud worms' can occur (Nell, 1993).

Estuarine and coastal systems differ in a number of respects, such as salinity regimes, temperature, hydrodynamics, flushing rates and hence residence time, phytoplankton composition and nutrient dynamics. Therefore, there are considerable differences in shellfish culture between these systems. Anthropogenic activities within the surrounding catchment area can also have a significant effect, and at times can have a major influence on the dynamics of these systems.

In recent times there has been much interest and concern with respect to the potential impact, or effect, of shellfish culture on the environment. Issues raised include problems with over-catch (e.g. settlement of Pacific oysters outside of marine farm areas - often referred to as feral oysters), biodeposition rates, impact on seagrass beds or reef communities and competition with, or displacement of, native species (pers. comm. Tony Thomas and Margaret Brett, DPIWE Marine Farming Branch). In addition, the question of carrying capacity is also frequently raised, defined here as the number of bivalves (e.g. oysters and/or mussels) which can be cultured in an area without detrimental effects on the ecology and natural functioning of a system. Additionally, shellfish farmers have expressed concern with respect to increased stocking rates,

extensions to leases, or applications for new leases within their area. For many of these changes, or new applications, concern has been raised by local shellfish farmers regarding the impact on, or degree of competition for, available food resources which may be experienced as a consequence. Their concerns are related to the effects of increased stocking rates and hence potential decline in individual farm performance, predominantly reflected in longer grow-out periods and time for oysters to 'come in' to condition.

The aim of this chapter is to provide an overview of processes and dynamics in systems without shellfish and compare them to those with cultured shellfish. A brief outline of the major components of system processes and dynamics will be discussed in the following sections. More detailed reviews will be provided in the relevant chapters following.

1.2 A brief background of phytoplankton dynamics, hydrological factors, nutrients and particulate matter

1.2.1 Water column processes

Firstly, it is important to consider the dynamics of phytoplankton growth and regeneration as this is a major source of food for shellfish. Phytoplankton growth is dependent on nutrients, light and temperature (Fig. 1.1). Sources of phytoplankton are either generation within an embayment (autochthonous) or imports from marine sources (allochthonous) via tidal exchange. The degree of zooplankton grazing can also reduce the available supply for shellfish, as well as populations of other filter feeders. Blooms of phytoplankton occur under favourable conditions of light, temperature and available nutrient supply, and there are often subsequent increases in zooplankton (Wassmann, 1991). If there is a rapid cycling of nutrients, high phytoplankton growth rate may be maintained (Kennish, 1990).

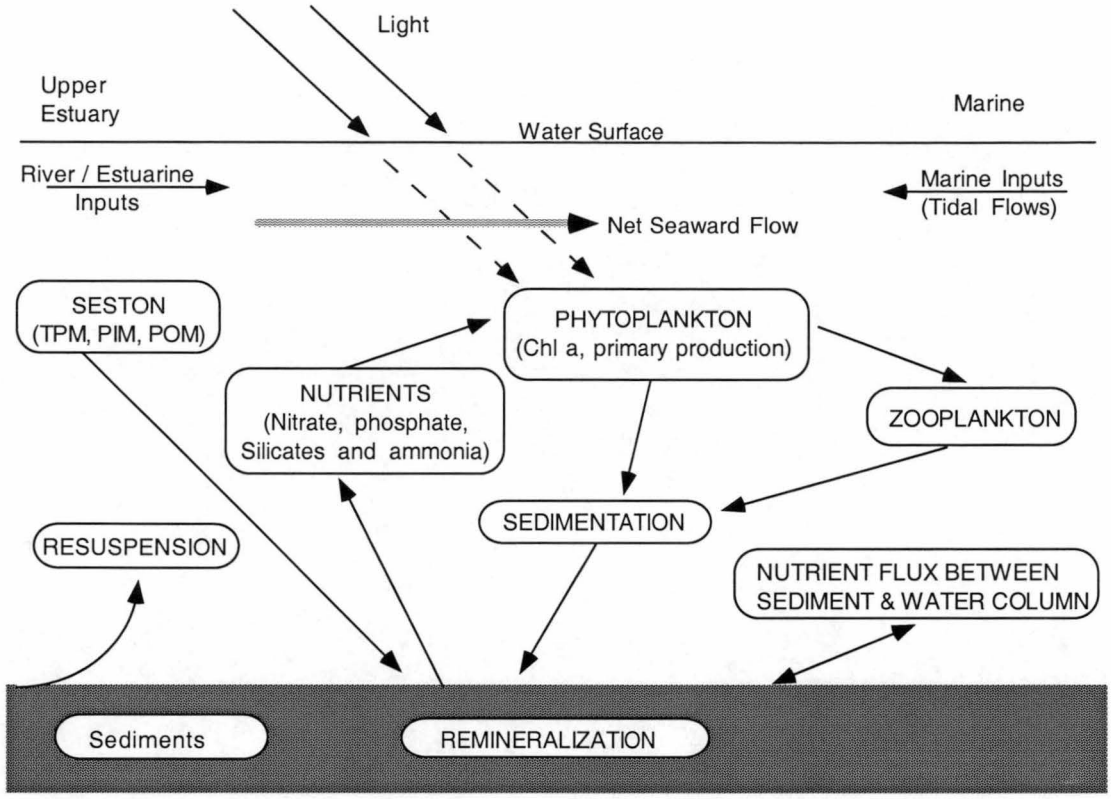

Fig. 1.1 A simple schematic diagram of the major phytoplankton and nutrient processes which occur within the water column and the sediment surface layer.

Phytoplankton are autotrophs dependent on light for photosynthesis. Chlorophyll a is frequently used as a measure of phytoplankton biomass since all algae contain chlorophyll a and some accessory pigments depending on species type (Parsons et al., 1977; APHA 1995). Availability of light is an important factor influencing species type and abundance (Parsons et al., 1977). Depth of light penetration through the water column is often reduced in estuaries, because of suspended matter (turbidity) caused either by riverine inputs, resuspension by wave action (mixing) or self-shading which may occur during dense algal blooms. It is often assumed that estuaries are highly productive, but it has been shown with some regions that turbidity, rather than nutrient limitation, has been responsible for reduced primary production (Joint and Pomroy, 1981; Cloern, 1987; Pennock and Sharp, 1994). Suspended sediments in the water column reduce light attenuation resulting in the photic zone being greatly reduced. Thus, though nutrients may be in abundance, which is often the case in estuarine systems, light limitation may be the major control over phytoplankton production (Cloern, 1987).

It has been shown that phytoplankton can adapt to turbid environments, with water column mixing moving cells up into the upper water column region and hence exposing

them to light, thus they are subjected to changing photosynthesis-respiration cyclic behaviour (Demers and Legendre, 1981; Fichez et al., 1992). Cole and Cloern (1987) reported that phytoplankton productivity in estuaries over periods of weeks to years can be estimated using phytoplankton biomass (mg m^{-3} chlorophyll a) and light availability. Though these authors stated that this procedure was only applicable where nutrients did not limit production.

Phytoplankton biomass in estuaries exhibits marked spatial and temporal variations (Jassby et al., 1997). Variations occur as the result of fluctuations in available nutrients, salinity, temperature, light, wind, water column mixing, water depth, fresh water inflows and current flows (Therriault and Platt, 1981; Jassby and Powell, 1994). Thus, phytoplankton can have a patchy distribution which may affect the uniformity of measurements of chlorophyll a from one area to another within an estuary or embayment. Also, dominant species vary according to conditions: diatoms tend to dominate in more turbid environments within estuaries or shallow water embayments, whereas flagellates tend to dominate in more settled waters (calmer, deeper). Benthic microalgal biomass has also been reported as an important contributor to the primary productivity of shallow systems (Lukatelich and McComb, 1986; Barranguet, 1997; Guarini et al., 1998).

Seasonal variations in phytoplankton occur, with the general trend of biomass tending to increase in spring followed by a decline in summer and secondary peak in autumn. Phytoplankton ecology is a complex dynamic process with many factors controlling species types, growth rates, condition, species successions, predation and regeneration (see Sommer, 1989) with quite diverse assemblages existing within a given body of water.

Water flow and circulation is an important influencing factor on phytoplankton biomass, with often displacement, due to a net seaward flow in estuaries, posing a restriction on phytoplankton growth rates. Thus, water residence time is an important consideration with respect to population growth in estuaries. *In situ* rates of production can be reduced, when rates of physical removal are high (Kennish, 1990). Areas with a short residence time generally contain species introduced from near shore oceanic waters (tidal inputs), whereas systems with longer residence times generate populations characteristic of the conditions prevailing within the estuary (e.g. Pinckney et al., 1997).

Phytoplankton reproduce asexually with each cell dividing in two once a day, though cell division may be faster if favourable conditions prevail (Tett, 1987).

Salinity can influence phytoplankton due to osmotic variations between freshwater and marine species. However, many phytoplankton species show considerable tolerance to salinity fluctuations, with adaptations occurring via cellular changes in ionic composition and hence osmoregulatory capacity. This has been shown with freshwater species flowing into coastal regions and vice versa (Kennish, 1990; Costanza et al., 1993). Temperature also influences growth and regeneration rate with variations between algal species.

An essential requirement for phytoplankton growth is the supply and availability of nutrients with nitrogen, phosphorus and silicon the most important. The forms of nitrogen preferentially utilised by most algae are ammonium, nitrate and nitrite which are predominantly used for cellular structural components. Ortho-phosphate is the predominant form of phosphorus and is a key requirement for cellular metabolism, and silicon is an essential nutrient for the external cell structure of diatoms (Riley and Chester, 1971; Parsons et al., 1977). Many diatoms have quite intricate and ornate external formations created by the deposition of hydrated silica. It is often reported that nitrogen limits growth in estuaries and near coastal systems (Probyn, 1992), however phosphorus limitation does occur (Nowicki and Nixon, 1985; Kimmerer et al., 1993; Pennock and Sharp, 1994).

Seston (which is often referred to as particulate or suspended matter) is a complex of living and non-living material. Often this comprises aggregates of detritus with associated microorganisms (bacteria) attached, which utilise the detrital components for cellular metabolism with resultant release of nutrients into the water column. The degree with which this occurs depends on the labile (readily utilisable) and refractory nature of the detritus. Phytoplankton (living and dead) are also important components of seston. It is difficult to determine the exact composition of the living and non-living components of seston, and often for simplicity only the organic and inorganic fractions are determined. Similarly to phytoplankton, considerable spatial and temporal variability can occur with seston concentrations (Berg and Newell, 1986; Baird et al., 1987).

Variations in seston concentrations can occur with sedimentation of this material, transport, flocculation and disaggregation, composition and stage of decomposition.

However, one of the most important aspects of seston, which is often overlooked or not recognised, is the bacterial component which plays a key role in water column nutrient dynamics. Bacteria within the water column often facilitate the aggregation of particles, by release of mucus substances which assist binding of this material, this process (or phenomena) has been referred to as 'marine snow'. It has only been in recent times that greater awareness and knowledge has been gained of the significance and importance of this component of water column nutrient dynamics (e.g. Cox, 1994).

1.2.2 Sediment processes

In shallow coastal waters and estuaries, nutrient regeneration (or remineralization) in the sediments from sedimented phytoplankton and organic matter has been shown to be important, with studies showing that a significant amount of nitrogen requirements for phytoplankton blooms can be supplied from the sediment (Forès et al., 1994). Nitrogen is considered to be a limiting nutrient for marine primary productivity, and hence its cycling is an important factor in the regulation of primary productivity (Blackburn, 1986). Recycling of phosphorus and silicon from sediments is also an important source to the overlying water (Feuillet-Girard et al., 1997).

In most estuaries autotrophic processes in the water column depend in part on nutrients recycled from benthic metabolism (Heip et al., 1995). Heterotrophic activities, which are concentrated at the sediment surface (or benthic boundary layer), depend on the deposition of particulate organic matter from the overlying water. Nutrient recycling from the benthos generally is sufficient to support 50% (or greater) of the primary production in the overlying water (Hopkinson and Wetzel, 1982; Kelly and Nixon, 1984; Dollar et al., 1991).

Increased loading of nutrients to coastal ecosystems generally results in elevated primary production and hence deposition of organic matter (sedimentation) which can, in turn, lead to depletion of oxygen from bottom waters. The processes of ammonification and nitrification increase with greater organic matter inputs, as long as there is sufficient oxygen (Fry, 1987; Brock and Madigan, 1991). If increases in organic matter are accompanied by a lowering of oxygen, creating anaerobic conditions, nitrification is inhibited (ammonium recycling reduced) and denitrification enhanced with subsequent loss of biologically available nitrogen (Brock and Madigan, 1991). Excessive loadings of organic matter can cause dramatic shifts in microbial metabolism and changes in

benthic infaunal assemblages (e.g. Kaspar et al., 1985; Castel et al., 1989; Grant et al., 1995). The upper layer, or surface, of sediments becomes anoxic, characterised by a black colouration and strong hydrogen sulphide odour, and in extreme cases azoic. Sediment type is also an important consideration, generally higher bacterial biomass occurs with finer silt/clay sediments than coarser sand sediments, attributed to the greater surface area: volume ratio (Fry, 1987; Kennish, 1990). Additionally, silt/clay sediments generally have a higher organic matter content as a result of their greater binding capacity. Generally, finer sediments occur in estuaries, particularly the upper reaches, with coarser sediments near the mouth (Day, 1981a).

Oxygen flux across the sediment-water interface is dependent on water flow across the sediment, with oxygen consumption influenced predominantly by sediment organic matter content, microbial and benthic faunal biomass and activity (Blackburn, 1987; Forster et al., 1996). Thus current flows, tidal flows or wind-forced mixing of the water column mass are important in supplying oxygen to the sediments. Depth of water, and hence light penetration, is also an important factor with respect to oxygen production by benthic algae.

Mineralization of organic matter, particularly in the upper sediment layer, causes nutrient enrichment of the interstitial pore water relative to the overlying water, with movement of nutrients by mixing processes or diffusive gradients (Rutgers Van Der Loeff, 1980; Simon, 1988). Sediment type and temperature have been found to be important factors influencing rates of benthic nutrient remineralization (Nowicki and Nixon, 1985). These authors reported greater rates of ammonium release from mud sediments as compared to those from sandy sediments, with sandy sediments tending to take up nitrate and phosphate. Greater diffusive flux rates of nitrate than ammonium have been shown in shallow sandy sediments (Simon, 1988), with the direction of ammonium fluxes related to water column stability (Simon, 1989). Ammonium fluxes into the sediment were found when sediment resuspension occurred (during the sampling period) and into the water column during calm periods (Simon, 1988).

Microbial processes within the sediment, and the water column, play a key role in nutrient recycling (Billen, 1978; Malone et al., 1986). The activities of the benthic infaunal community also contribute, via bioturbation, bioirrigation and grazing/predation of the microflora and organic matter/detritus (Graf and Rosenberg, 1997). Microphytobenthic biomass has also been found to be significant in not only

producing oxygen (Barranguet, 1997), but contributing to water column productivity and nutrient cycling (Lukatelich and McComb, 1986; Barranguet et al., 1994; Feuillet-Girard et al., 1997; Guarini et al., 1998). Highest biomass was found on mudflats (Guarini et al., 1998) and shallow coarse sandy sites (Lukatelich and McComb, 1986) with light availability an important factor on distribution.

1.3 Oysters - filtration, food requirements, growth, biodeposition and nutrients

Oysters are filter feeders which remove material from the water column as their source of food by a pumping and filtering mechanism (Dame, 1996). Detailed descriptions of the anatomy and physiology of oysters can be found in, for example, Winter (1978), Quayle and Newkirk (1989) and Dame (1996). Briefly, water is pumped through the gills and intercepted particles are sieved and selected. Particles are either transported to the mouth, or bound in mucous and rejected as pseudofaeces. Particles ingested are eventually discharged as faeces. The consistency of these two forms of excretion are reasonably distinct; faeces are more compact 'ribbons', whereas pseudofaeces are more loosely compacted (often described as 'fluffy'). Deposition of faeces and pseudofaeces to the sediments makes the biodeposits available to the benthos. Nutrients are consequently released to the water column and hence available to phytoplankton (Fig. 1.2). Assimilation of material into oyster tissue does remove elements from the system (e.g. carbon, nitrogen, phosphorus) (Dame et al., 1989), though some direct nutrient input to the overlying water occurs from oyster excretions (ammonia, urea, amino acids) (Dame et al., 1985; Boucher and Boucher-Rodoni, 1988).

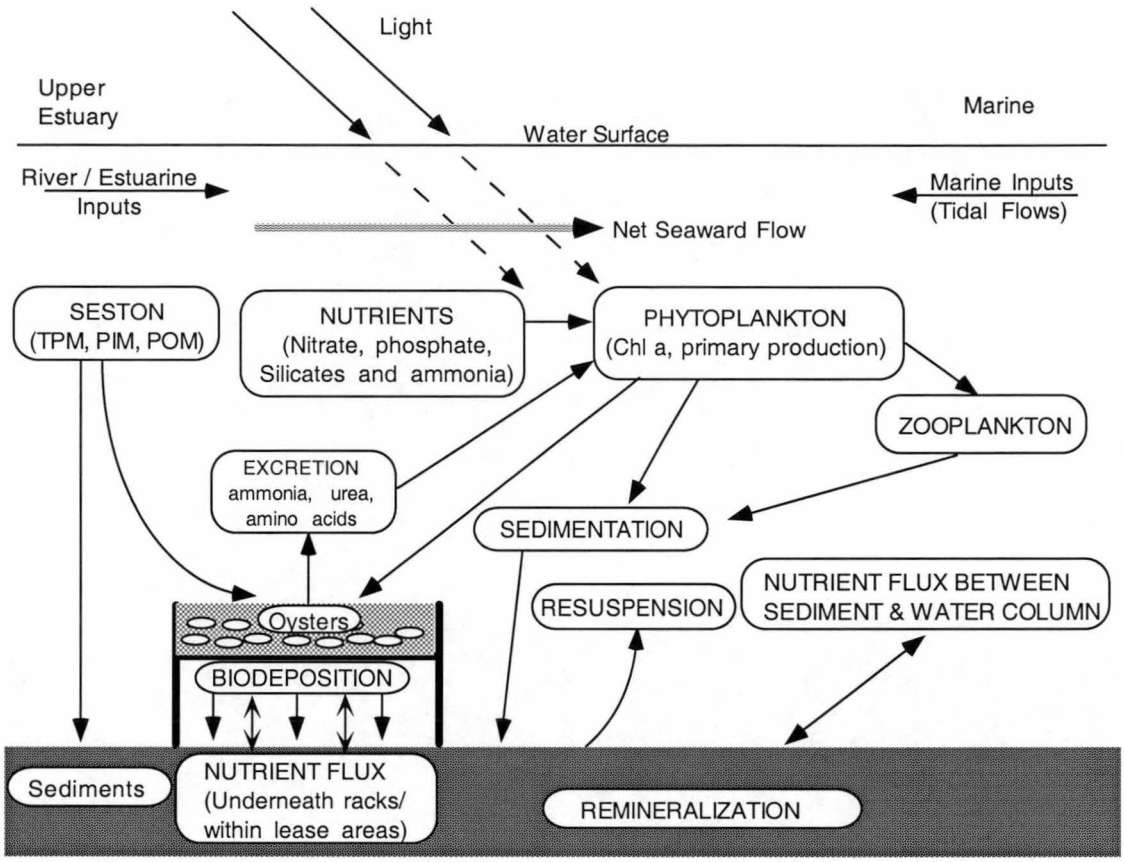

Fig. 1.2 A simple schematic diagram of the major processes of oysters, phytoplankton, seston and nutrients which occur within the water column and sediment.

1.3.1 Particle removal

Removal of different sized particles is variable because of preferential uptake (selectivity), and is dependent upon the predominant size fractions available and seston quality (Kiørboe and Møhlenberg, 1981; Shumway et al., 1985; Prins et al., 1994; Pastoureaud et al., 1996; Bayne, 1998; Grant and Bacher, 1998; Hawkins et al., 1998). Shellfish can vary their filtration rate according to the composition and concentration of seston, with rejection of material as pseudofaeces above certain thresholds (Widdows et al., 1979; Barille and Prou, 1993; Hawkins et al., 1998). However, whilst it is generally regarded that the inorganic fraction of seston has a diluting effect on the organic fraction, studies have shown that not all material ingested is composed of organic matter, with inorganic matter (e.g. silt) comprising a significant and preferential component of ingested material (Kiørboe et al., 1981; Bayne et al., 1987; Hawkins et al., 1996). Oysters can filter bacteria, organic and inorganic matter, phytoplankton and

zooplankton with size ranges reported within the range of 0.8 - 146 μm (Haven and Morales-Alamo, 1966).

The ability of oysters to capture seston from the water column has been found to vary in response to current speed, particle concentration and composition (Walne, 1972; Lenihan et al., 1996; Hawkins et al., 1996; Grant and Bacher, 1998). *In situ* clearance rates of 5-7 and 2.5 L h^{-1}/standard oyster and mussel respectively were measured by Smaal and Zurburg (1997) who, in contrast, found no relation between clearance rates and current velocities. These authors surmised that filtration rates were determined by seston and chlorophyll concentrations only. It was also estimated in this study that bivalves within the Marennes-Oléron Bay (France) had the potential to filter the total bay volume within 2.8 days, based on an average clearance rate of 1.8 $m^3m^{-2}d^{-1}$. Given that the residence time of water within the bay was calculated to be 7 days, it was obvious that generation of particulate material (food) must have been occurring within the bay. In fact Smaal and Zurburg (1997) hypothesised that microphytobenthos (predominantly benthic diatoms) generated within the bay, were an important source of food to these bivalves, with nutrient remineralization from biodeposits playing an important role in this process.

Particle removal is also variable according to density of bivalves and orientation to prevailing water flow. This has been found with mussel raft culture in the Ria de Arosa (Spain) where mussels on the front of rafts (orientated towards the current) reduced seston concentration by 24% greater than those mussels located at the rear of the raft (Navarro et al., 1991). Similarly, Heasman et al. (1998) found that local limitation of food supply occurred within mussel rafts in Saldanha Bay (South Africa), and this was influenced by row spacing, stocking density and age of mussels. In this same system, Boyd and Heasman (1998) showed that current flows within and around mussel rafts varied according to design (row spacing and size) and distance between rafts within a farm and between farms. Rodhouse et al. (1985) showed that mussel rafts in Killary Harbour (Ireland) cleared approximately 50% of particulates from the water column, the predominant food source in this system being phytoplankton.

A number of *in situ* studies have been done on oyster reefs and mussel beds using enclosed tunnels covering sections of these systems (Dame, 1987; Dame and Dankers, 1988; Dame et al., 1989; Dame et al., 1991; Asmus and Asmus, 1991; Dame et al., 1992). Uptake and release of materials have been studied over tidal cycles by measuring

differences between the inflow and outflow water. Such *in situ* assessments of bivalve dynamics have provided more realistic information, as the natural community structures associated with these reefs or beds are also involved. This includes the sediments, associated bacteria, benthic infauna, benthic phytoplankton and water flow movements. Large quantities of phytoplankton (measured as chlorophyll a) removal have been recorded for oyster reefs (Dame et al., 1992) and mussel beds (Dame and Dankers, 1988) indicating that these systems can considerably influence water column productivity through grazing (biofiltration). Various concurrent observations have shown significant releases of ammonium and ortho-phosphate from bivalve metabolism, which become available for further phytoplankton growth, thus establishing a feedback system. Bacteria in the sediments under bivalves may also release ammonium as a byproduct from the decomposition of sedimented organic matter, and this ammonium is available for phytoplankton production to replenish stocks grazed by the bivalves (Dame et al., 1992). Boucher and Boucher-Rodoni (1988) concluded that of the calculated nitrogen fluxes for an oyster reef, 40% released was due to oysters with the rest from the sediments. Thus, bivalves are considered important components in the nitrogen cycles of estuarine and coastal systems (Lerat et al., 1985; Liu and Fang, 1986; Grenz et al., 1991; Barranguet et al., 1994).

Oyster reefs release large quantities of inorganic nutrients into the water column, which may then enhance primary production. A study by Dame and Libes (1993) on two similar tidal creek systems, one with oysters and the other with oysters removed, showed no significant differences in chlorophyll a concentrations between the two. They concluded that despite the grazing pressure from the oysters filter feeding, the remineralization of nitrogen and phosphorus metabolic by-products kept nutrient levels at the same level as water-flushing rates. That is, they were able to maintain phytoplankton levels equivalent to the levels in the tidal creek system without oysters.

Net flux of inorganic particulate material measured over an oyster reef showed a general trend of net uptake on flooding tides and usually net release on ebbing tides (Dame, 1987). This trend was assumed to be the result of higher water velocities on ebbing tides with resuspension of particles occurring. Tidal variation in seston has also been reported by Fegley et al. (1992), who found that seston differed before and after slack low water. Low quantities of highly variable seston were replaced by higher quantities of consistent quality seston during the flood tide, with variations in seston between tides almost as great as those measured within a year. Depletion of food (measured as

chlorophyll a) in the waters above mussel beds has been shown, with water movement a critical factor in supply of food to the mussels (Fréchette and Bourget, 1985a; Fréchette and Bourget, 1985b). Fréchette and Grant (1991) showed that wind-driven resuspension did not mitigate this process, with phytoplankton depletion the main causal factor for reduced growth in mussels on the bottom as compared to those held 1 m above.

Decline in oyster (*Crassostrea virginica*) stocks in Chesapeake Bay has been implicated in changes to the ecology of this bay system, with increased primary production and anoxic conditions in bottom waters during the summer periods (Newell, 1988). *C. virginica* is highly efficient at removing suspended particulate matter >3 μm, assimilating approximately 70% of this material. Newell (1988) suggests that the decline in oyster biomass has reduced grazing of phytoplankton, with resultant anoxic conditions occurring at times in summer following the sedimentation of these phytoplankton blooms. In Chesapeake Bay, in which numerous and extensive studies have been performed, it has been hypothesised that oysters and other bivalve filter-feeders have been an important component of the ecosystem of this bay. In light of oysters being effective controllers of phytoplankton biomass, Newell (1988) estimated that pre-1870 oyster stocks would have been capable of removing 77% of the 1982 daily carbon production in waters less than 9 m deep. This has led to the strong belief that oysters and other benthic filter feeders were effective controllers of primary productivity and water quality in Chesapeake Bay (Ulanowicz and Tuttle, 1992). One of the major problems with Chesapeake Bay currently, is the predominance of phytoplankton which has a significant consequence on the ecosystem. The general philosophy has been to re-introduce oyster populations into the bay as a means of controlling these phytoplankton populations and hence rehabilitating the bay system (Gottlieb and Schweighofer, 1996).

1.3.2 Biodeposition

Unlike fish farms where fish are fed with external inputs, shellfish are reliant on suspended material in the water column as their source of food. Thus, additional external loadings are not placed on the ecosystem. However, in the process of filtration of this material, changes do occur in the size spectrum and consistency of the natural particulate material biodeposited (Haven and Morales-Alamo, 1966; Bernard, 1974; Kautsky and Evans, 1987; Grenz et al., 1991) and hence sediment texture (Kaspar et al., 1985; Grant et al., 1995). Biodeposits consist of two fractions, faeces and pseudofaeces,

and the amount and type of faecal or pseudofaecal production is dependent on the quantity and quality of the ambient seston. Pseudofaeces, which essentially are composed of uneaten/rejected suspended particles bound in mucus, break down more readily than intact faeces (Iwama, 1991).

Sedimentation of particulate material from bivalves can either have a physical effect due to coverage and build-up on the bottom (particularly for areas with poor current flow) (Grant et al., 1995), or chemical effect brought about by remineralization of this material predominantly by the bacterial and benthic faunal community (Bernard, 1974; Sornin et al., 1983; Boucher and Boucher-Rodoni, 1988; Barranguet et al., 1994; Grant et al., 1995). Utilisation of sedimented material by the benthos varies according to the number and types of microorganisms and benthic fauna present in the sediments, as well as the current flow (supply of oxygen to the sediments and benthic boundary layer). Obviously, those areas with good current flow and rich benthic community will degrade sedimented material much more efficiently and effectively than areas which lack these features. The nature of biodeposits will also affect the rate of sedimentation and remineralization, with current flow affecting their transport in the water column.

The amount of biodeposition (faeces/pseudofaeces) by shellfish can be significant (Haven and Morales-Alamo, 1966; Bernard, 1974; Feuillet-Girard et al., 1988). Laboratory studies conducted by Haven and Morales-Alamo (1966) indicated that oysters on 0.405 ha of an estuarine bottom may produce up to 981 kg of biodeposits weekly. Deposition rate was influenced by temperature, with reduced rates recorded at low water temperature (6.7° C) and very little deposits measured at 2.8° C (Haven and Morales-Alamo, 1966). These authors also showed that the size range of particles in the faeces and pseudofaeces ranged from 0.8-146 μm, reflecting the natural particles and phytoplankton types observed in the York River from where the water in which the oysters were held was obtained.

Sornin et al. (1983) showed that the quantity of biodeposits changes seasonally, with variation mainly attributed to the turbidity of the water. A positive correlation ($r = 0.76$) was found between average seston in the water and quantity of biodeposits. The rate of pseudofaeces production increased proportionally to the turbidity of the water, with reduced rates occurring at very high turbidity, indicating that oysters have difficultly living in turbid water (Sornin et al., 1983).

1.3.3 Mineralization

Oysters have been regarded as effective controllers of turbidity by removing particulate organic and inorganic matter from the water column and depositing this material on the sediments (Newell, 1988). These biodeposits are more resistant to erosion and resuspension than naturally sedimented material (Widdows et al., 1998). Mussels have been shown to increase the sedimentation rate of small particles through biofiltration, increasing the quality of the sediments by addition of organic matter (Dame et al., 1991). This enriched sediment results in increased microbial and benthic biomass and activity with nutrient recycling, from nutrient regeneration, occurring as a consequence. However, increases in microbial activity may lead to sulphide production following increased oxygen consumption during the degradation of organic matter when insufficient oxygen is available (Dahlbäck and Gunnarsson, 1981). Such a process occurs when sediment loading is excessive, with the degradation rate being exceeded by supply of organic matter. Poor current flows would also exacerbate this process, as rapid depletion of oxygen would occur - current flow providing a means of supplying/replenishing oxygen levels at the benthic boundary layer and also acting to assist diffusion between the water column and benthic boundary layer.

Increased sedimentation under shellfish culture as a result of biodeposition has been shown to increase nutrient concentrations and fluxes (Kaspar et al., 1985; Liu and Fang, 1986; Boucher and Boucher–Rodoni, 1988; Grenz et al., 1991). Many studies have shown that the predominant nutrient released from sediments associated with shellfish is ammonium, with higher concentrations released under shellfish culture as compared to control sites away from farms (Sornin et al., 1983; Lerat et al., 1985; Kaspar et al., 1985; Dame et al., 1985; Feuillet–Girard et al., 1988; Boucher and Boucher-Rodoni, 1988; Sornin et al., 1990; Barranguet et al., 1994; Hatcher et al., 1994; Grant et al., 1995; Smaal and Zurburg, 1997). A number of these studies also showed greater accumulation of organic matter in the sediments from biodeposition. Measurements of ammonium release were found to be higher under mussel farms as compared to oysters (Barranguet et al., 1994). These authors attributed this difference to factors such as higher biomass of mussels per unit area, differences in biodeposit composition and differences in the size of cultivation areas of oysters and mussels in their particular study site.

Benthic nutrient fluxes studied in a Penghu Bay oyster farm showed that nutrient regeneration from the sediments, stemming from organic input, acted as a source for primary productivity within this system. Movement of nutrients, predominantly ammonium, followed diffusive gradients between the sediments and the water column (Liu and Fang, 1986). A study of sediments within an intensive oyster growing area in the Bay of Morlaix (France), showed a strong accumulation of nitrate and ammonium in sediments at the end of summer and winter, but reduced accumulation in autumn (Lerat et al., 1985). No transfer of nitrate was observed from the sediment to the water column, however transfer of ammonium occurred with direction to/from the sediment dependent on the concentration of ammonium in the water column. Boucher and Boucher–Rodoni (1988) noted that nitrate was mainly absorbed from the water column into the sediment, while Lerat et al. (1985) suggested that sediment nitrate tends to be transformed to ammonium or denitrified. Similar trends of accumulation or release have been shown with ortho-phosphate and silicates, but flux rates and concentrations were considerably less than those measured for ammonium (Kaspar et al., 1985; Sornin et al., 1986; Dame et al., 1991; Grenz et al., 1991; Feuillet-Girard et al., 1997).

Recent sedimentation of fresh organic matter has been shown to be rapidly remineralized, enhancing nutrient release to the overlying water column with increased flux rates observed (Kelly and Nixon, 1984; Grenz et al., 1991). However, direction of flux of nutrients (that is, to or from the sediments) does not always follow diffusive gradients, with factors such as temperature, microbial biomass and activity, oxygen availability, water immersion period and water column mixing influencing the uptake and/or release of nutrients (Simon, 1988; Feuillet-Girard et al., 1997).

Microphytobenthic biomass has been shown to be higher under shellfish culture sites compared to control sites, contributing to the process of nutrient flux from biodeposits (Barranguet et al., 1994; Feuillet-Girard et al., 1997; Smaal and Zurburg, 1997; Barranguet, 1997). Analysis of phytoplankton pigments in sediments under shellfish culture have generally shown considerably higher levels of phytoplankton pigments (chlorophyll a or phaeopigment (degraded chlorophyll a)) concentrations as compared to control sites. Barranguet et al. (1994) and Barranguet (1997) found chlorophyll a concentrations to be 2–3 times greater under oyster or mussel cultures than those measured at reference sites. Similarly, Dahlback and Gunnarsson (1987) found phaeopigment concentrations to be 10 times greater under culture sites, whilst Kaspar et al. (1985) found an approximate 3 fold increase. These differences were largely

attributed to shellfish biodeposition, either as a consequence of ingestion of phytoplankton, resulting in degraded chlorophyll products, or promotion of microphytobenthos due to sediment nutrient production as a result of biodeposits. Seasonal variation in sediment chlorophyll a levels, both at reference sites and under shellfish culture, were principally attributed to variations in seston phytoplankton abundance, temperature and feeding activity of the shellfish (e.g. Kaspar et al., 1985; Barranguet, 1997).

Microphytobenthic algae thus have a mitigating effect on nutrients released from the sediments, particularly under shellfish culture, by utilising these for their own production with subsequent increase in oxygen production during photosynthesis. However, benthic oxygen flux studies have shown that net oxygen fluxes are mainly negative under intensive high density shellfish culture with degradation of organic matter principally by anaerobic processes (Boucher and Boucher–Rodoni, 1988; Barranguet, 1997), though Boucher and Boucher–Rodoni (1988) showed that occasional aerobic degradation (nitrification) occurred. Barranguet (1997) noted that whilst microphytobenthos production under a mussel farm in the Mediterranean was higher than at a reference site, the negative net oxygen fluxes indicated that the microphytobenthos could not supply the sediment oxygen demand.

1.4 Linking water column parameters with oyster growth and carrying capacities

A question which is most frequently raised and increasingly of concern with respect to shellfish growing areas is: what is the carrying capacity?. That is, how many shellfish can be grown in an area, what are appropriate stocking rates, what are the impacts on the ecology of an area, and issues regarding the sustainability of culture operations. These are asked by industry, government managers and regulators, and members of the public.

Studies on the effects of shellfish culture have generally focused on water column and sediment processes. Dense cultivations, or accumulations, of filter feeders can result in detrimental effects, such as reduced growth rates, poor condition or longer grow-out times and hence reduced production per annum within and between farms in a given water body, as has been shown in Japan, France and Spain (Rosenthal, 1994a). Reduced production has been attributed to competition for available food resources. Intensive

culture of shellfish can also place strong competition on available food resources for natural filter feeders. This can potentially lead to significant detrimental effects on the ecology of a system.

The carrying capacity of an area has been defined as "the stock density at which production levels are maximised without negatively affecting growth rates" (Carver and Mallet, 1990). However, a factor which needs clarification, is at what point is 'negatively affected growth rates' defined? It is assumed this is when production levels decline and it takes longer for shellfish to reach market size and condition relative to past experience within an area. Grant et al. (1993) define carrying capacity as "the number of bivalves which can be sustained at a specified growth rate". Another consideration when determining the carrying capacity of an area is, what percentage is apportioned, or recognition given, to other filter, or detrital, feeders in the system. This could be defined as the 'ecological allocation'. This is an important consideration, not only to satisfy a compromise between the wider community and industry, but more importantly to ensure the healthy functioning of natural system processes and dynamics.

The culture of shellfish is dependent on the production and supply of phytoplankton and other food sources, its consumption by these filter feeders, and its transformation into body tissue and hence growth (Grant, 1996). Determination of a figure for the carrying capacity of a given growing area is not simple, and is complicated by natural temporal and spatial variations which occur on much longer time scales than the duration of studies performed to determine carrying capacity. The preceding sections have discussed some aspects related to what needs to be measured, or considered, in order to calculate carrying capacity. Briefly, these include 1) the type of food available and its production or regeneration, 2) particle removal (i.e. filtration) and factors which control selectivity and consumption, and 3) the supply, or transport of food, namely the hydrodynamic characteristics. Measurement of bivalve condition and growth rate is also an important factor to assess whether the carrying capacity has been exceeded.

Approaches to the question of carrying capacities entail a number of different methods from empirical studies, calculation of budgets and simulation modelling (Grant et al., 1993). Fundamentally, all studies involve the determination of bivalve growth based on food and temperature which are seen as the two most important factors which regulate bivalve growth (Brown and Hartwick, 1988a; Roland and Brown, 1990; Hickman et al., 1991; Hofmann et al., 1992). Empirical studies are largely based on correlation between

growth and single or multiple environmental factors, such as temperature, chlorophyll a, particulate organic matter (Brown and Hartwick, 1988a; Carver and Mallet, 1990; Hickman et al., 1991; Grant et al., 1993; van Stralen and Dijkema, 1994). Another approach has been via the calculation of budgets to determine phytoplankton biomass and the ingestion requirements of bivalves over various time scales (from daily, seasonal or yearly) to calculate the biomass of shellfish which can be sustained in a given area (Grant et al., 1993). This approach is based on the development of nutrient budgets calculated from flux measurements of carbon, nitrogen and phosphorus linked to tidal hydrologic models to calculate daily net inputs to, or removal from, the water column. One such example of this is the study of mussels in Killary Harbour (Ireland). Rodhouse et al. (1985) calculated that for 1 m^2 of mussels suspended from a mussel raft in Killary Harbour, 55.53 kg m^{-2} are consumed with 8.13 kg m^{-2} (mussels) harvested, assuming a carbon content of live mussels to be 2.95%.

A third approach which is increasingly becoming of interest to researchers, are simulation models (Raillard and Ménesguen, 1994; Hawkins et al., 1998; Grant and Bacher, 1998). These consist of various compartment sub-models of processes, such as consumption (filtration), in relation to factors which drive these systems such as temperature, or food concentrations. However, it has been recognised that there is still a considerable lack of understanding of shellfish physiology and behaviour, especially with respect to feeding (Bayne, 1998). A workshop held at the Plymouth Marine Laboratory (England) in October 1996 explored aspects of suspension feeding processes and how these related to, or could be used in, carrying capacity studies (Bayne, 1998). A summary of the outcomes of this workshop were reported in Crawford et al. (1996) where the participants concluded that whilst modelling was important in predicting bivalve growth rates, modelling of carrying capacity is very difficult because of the complications of different spatial and temporal scales involved. Spatial scales vary from mm for sediment organic matter and nutrient accumulation to kilometres for tidal water movements in estuaries: temporal scales vary from seconds and minutes for physiological responses to seasonal and annual variations in climatic factors (Crawford et al., 1996). Bayne (1998) reported that much additional information is still required on physiological parameters to be incorporated into carrying capacity models. Briefly, these were: modelling feeding rates, quantification of particle removal rates, modelling phytoplankton growth, and shellfish growth rates.

Doering and Oviatt (1986) have cautioned the use of laboratory based filtration rate model results for application to carrying capacity studies, as these yield poor predictions of sedimentation and phytoplankton removal in natural systems. More appropriate filtration rate models are those obtained from using natural seston suspensions. Another approach to assessment of feeding and absorption is the biodeposition method, whereby measurement is made of the suspended particles and biodeposit production (Iglesias et al., 1998). While these authors noted factors which need to be taken into account with this method, such as the time lag between suspended particles ingested and biodeposit production, as well as quantitative and separate collection of faeces and pseudofaeces, the method does provide reasonable estimates of clearance rates and scope for growth predictions.

Dynamic models which incorporate hydrodynamic information are seen as the preferred option, as these permit estimations from various scenarios. They also take into consideration the transport and supply of food material to shellfish. More complex models could be used to recommend appropriate spatial configurations of shellfish culture within, or between, farms to maximise production based on carrying capacity estimates.

Field sampling and the collection of hydrological data (current flow measurements, tidal movements, freshwater inflows) are invaluable for the development of models. Shellfish growth rates measured concurrently with water column parameters provide a valuable means of assessing environmental factors which affect growth and for comparison with model predictions. However, it is important to recognise that while carrying capacity estimates enable determination of shellfish farm production, consideration needs to be given to the whole system to ensure and maintain healthy ecosystem functioning (e.g. Baudinet et al., 1990; Simenstad and Fresh, 1995; De Casabianca et al., 1997). That is, a more ecological focus, or approach, should be adopted. This will not only ensure sustainable production for industry and maintenance of reasonable growth rate times, but will ensure maintenance of the ecological sustainability.

1.5 A brief history of oyster culture in Tasmania

The history of Pacific oyster (*Crassostrea gigas*) farming in Tasmania has been well documented by Sumner (1972; 1974). Briefly, the industry stemmed from oysters

introduced from Japan in the late 1940s and '50s. Originally the first site for introduced oysters was in Upper Pitt Water by CSIRO Division of Fisheries - though the oysters survived, spat fall was low. The oysters were then transferred to Port Sorell, an estuary which was considered to have more favourable conditions for growth and spawning and by the late '50s a number of mass spawnings and heavy spat falls were recorded (Thomson, 1958; Sumner, 1974). During this time, Pacific oysters were observed within the lower reaches of the Tamar River estuary where the population flourished, though it remains uncertain as to how they arrived within this estuary. The early development of the oyster culture industry started in 1968 and was sustained by wild spat caught on tarred sticks within the Tamar River and on-grown on intertidal leases located elsewhere in Tasmania.

Unreliable spat catches in the late 1970s and increasing demand prompted the establishment of a hatchery at the Department of Sea Fisheries Marine Research Laboratories (Taroona) in 1978. Successful spawnings were achieved and commercial quantities of spat produced and sold to industry. Subsequently, two commercial hatcheries at Bicheno and Dunalley were established and began production of Pacific oyster spat. These are now the main supply of oyster seed to industry, following nursery growth at a number of locations around the state. Juvenile oysters are supplied to growers at approximately 6-8 mm for on-growing to harvest size (60-120 mm).

There are currently approximately 36 shellfish growing areas in Tasmania, most of which are located within relatively sheltered coastal embayments or estuaries. The predominant form of oyster culture in Tasmania is intertidal, where oysters are held in mesh bags or baskets suspended from wooden raking. Deep water farms employ sub-tidal culture techniques where stacks of mesh trays are suspended from buoyed horizontal long lines. During their growth, the oysters are regularly removed and graded to ensure uniform sizes are maintained within the mesh bags or baskets, and to alter densities within these containers as the oysters increase in size.

The industry has developed rapidly since the late 1970s. In 1977 there were 12 oyster farms producing 147 thousand dozen oysters increasing to 111 farms in 1996 producing approximately 5 million dozen (source DPIWE, Marine Farming Branch). Current value of the industry to the Tasmanian economy is estimated to be $15 million. The Tasmanian Pacific oyster industry has the potential for further expansion and already it is recognised as a significant primary industry and contributor to the Tasmanian

economy. Demand is continually increasing, hence there is pressure to increase lease sizes, stocking densities, or establish new leases. Suitable new areas are increasingly difficult to obtain and there has been strong objection from some members of the community to further leases within (or near) existing growing areas.

Applications for extensions to existing marine farms or new areas continue to be submitted to the DPIWE Marine Resources Division, the state regulatory authority for marine farm leases. Formerly such applications were processed individually in an ad hoc fashion. However, this process was fraught with difficulties including concerns raised by existing lease holders with respect to the degree of competition for available food resources with the granting of leases within their areas, issues of sustainability, carrying capacity, site suitability, environmental impacts and conflicts with other users of the areas (recreational, aesthetics, commercial). This led to the establishment of a zoning scheme for the planning and development of marine farming in Tasmania. Marine Farming Development Plans have been, or are being, prepared in accordance with the Marine Farming Planning Act 1995 for the main aquaculture regions of Tasmania. These plans identify areas of water that may be suitable for marine farming, with consideration of other users of the coastal zone (source DPIWE, Marine Farming Branch).

1.6 Objectives of study

The objectives of this study were to assess the relationships between water quality parameters, hydrodynamic characteristics and oyster growth in three major Pacific oyster (*Crassostrea gigas* Thunberg, 1793) farming areas in Southern Tasmania with the aim of determining reasons for differences in reported oyster growth rates and shellfish productivity between the areas.

The specific objectives were:

- To determine nutrient, phytoplankton biomass (measured as chlorophyll a) and seston quality and quantity in the water column at three oyster farming areas characteristic of different systems.
- To measure the water column parameters of temperature, salinity and secchi depth concurrently within each of the farming areas.

- To determine oyster biodeposition rates, quality and influence on the sediments under culture structures.
- To measure oyster growth rates and condition at representative sites within each of the three areas.
- To determine the hydrodynamic characteristics of each area with respect to volumes, flows, exchange rates and residence times.
- To examine the relationships between the environmental parameters and oyster growth rates and shellfish production.

2. Water quality parameters

2.1 Introduction

In Tasmania, most shellfish farms are located within estuaries or coastal embayments. The majority of these farms cultivate Pacific oysters (*Crassostrea gigas*) intertidally. Such locations are favoured since they generally provide extensive areas of intertidal sand/mud flats, are relatively sheltered, enable easy access to farm sites and often shore-based facilities can be located nearby.

The farming of shellfish has to a large extent been developed based on practical experience (Winter, 1978). For successful production to achieve optimal and sustainable conditions for growth, and to be able to predict future production, knowledge of environmental parameters such as food quantity and quality, salinity, temperature, and water flow are of great importance. Many studies have been conducted on oysters and mussels culture areas in Europe or America, but few studies have been performed within Tasmania, or Australia.

Estuaries and coastal embayments are complex dynamic systems, with many factors influencing the processes which occur within them, such as freshwater inputs, tidal flows and volumes, bathymetry, anthropogenic sources and inputs, flushing rates or residence times, prevailing weather conditions and biological factors (such as presence of seagrass beds, benthic invertebrate populations, mud flat and salt marsh areas). Most water column studies of these systems entail assessment of phytoplankton biomass and/or production, nutrients, particulate matter quantity and quality, temperature, salinity and degree of light attenuation or turbidity (Parsons et al., 1977). Considerable variability can occur temporally and spatially in these parameters, predominantly due to the hydrodynamic regime of the systems and seasonal influences. Approaches adopted and parameters measured are numerous. Selection of sample sites, sampling frequency, methods, and analytical procedures in which to adequately and comprehensively study such systems can entail much time, effort and expense.

For studies of shellfish culture areas, and in this case oyster farming areas, the key focus is on estimation of food availability and supply. Many studies have been conducted on existing shellfish growing areas with the aim of understanding the reasons for variation

in shellfish growth, or extrapolation of results to predict future estimations of biomass (Widdows et al., 1979; Rodhouse et al., 1984; Bradford et al., 1987; Carver and Mallet, 1990; Hickman et al., 1991; Ball et al., 1997; Pitcher and Calder, 1998).

Chlorophyll a is frequently used as a measure of phytoplankton biomass. However, this measure has limitations primarily because of variations in carbon/chlorophyll ratios which vary with species, light intensity, nutrient availability and physiological condition of the cells (Kennish, 1990). Particulate matter, often termed "seston", is also a considerable component of "food" available to shellfish (Berg and Newell, 1986). It is considered that the particulate organic matter (POM) fraction forms the main food source for oysters. However, studies have shown that the particulate inorganic matter (PIM) component can be utilised as part of oysters diet (Kiørboe et al., 1981; Bayne et al., 1987; Barillé and Prou, 1993; Hawkins et al., 1996; Bayne, 1998). Kiørboe et al. (1981) found the presence of suspended silt to have a stimulating effect on mussel growth and clearance rates. Studies of shellfish feeding behaviour have shown that selectivity of particles and alteration of filtration rates occurs to maximise available food resources (Kiørboe and Møhlenberg, 1981; Gerdes, 1983; Shumway et al., 1985; Pastoureaud et al., 1996; Hawkins et al., 1998).

Inorganic nutrients are measured as a means of assessing either the potential for phytoplankton growth, or often the limitation of growth due to insufficient quantities available. Nitrate, nitrite and to a lesser extent ammonium are analysed, with nitrate and nitrite preferred because the analysis of low level ammonium concentrations is difficult and fraught with complications. Often "nitrates" are reported as NOX, which is nitrate + nitrite. The major form of phosphorus utilised by phytoplankton and the one most frequently measured is the dissolved inorganic form, ortho-phosphate (Parsons et al., 1977; Kennish, 1990). Nitrogen and phosphorus are considered the two major nutrients required for phytoplankton growth and measured to assess not only sources, concentrations and seasonal trends but also to determine if phytoplankton production is limited by the availability of one of these.

It is assumed in marine systems that nitrogen is the nutrient most limiting, with phosphorus the limiting nutrient in freshwater systems (e.g. Day, 1981b). Assessment of this is conducted utilising the Redfield ratio where the N:P content of phytoplankton, 16:1, is compared to the N:P ratio of water samples either using total nitrogen (TN): total phosphorus (TP) or dissolved inorganic nitrogen (DIN): dissolved inorganic

phosphorus (DIP) (Parsons et al., 1977; Coughanowr, 1995; O'Donohue and Dennison, 1997). When this ratio is greater than 16, it is assumed that phosphorus limitation occurs, whilst with ratios <16 nitrogen limitation occurs. However, consideration needs to be given to other factors which may be responsible for limitations on phytoplankton biomass, such as light limitation (Joint and Pomroy, 1981; Cloern, 1987; Fichez et al., 1992). Additional studies which can be performed to ascertain limitation are laboratory conducted nutrient enrichment and light intensity experiments on water samples collected (e.g. O'Donohue and Dennison, 1997).

The measurement of silicon, in particular dissolved (or 'reactive') silicon, provides additional information on estuarine nutrient dynamics, as this is an important nutrient for diatoms which often form a considerable component of the phytoplankton population in shallow coastal embayments and estuaries.

Seasonal trends in nutrients occur, predominantly in response to biological uptake by phytoplankton, with the timing and growth of phytoplankton largely dependent on temperature and light. A trend observed is increased nitrates with reduced chlorophyll a concentrations (Kennish, 1990). The reverse happens when conditions favourable to phytoplankton growth occur and the increase in biomass, shown with increasing chlorophyll a levels, corresponds to a reduction in nitrates. However, the nitrogen cycle in estuaries is complex, and regeneration of nitrogen sources via remineralization by the benthos may be occurring to compensate for losses in the water column.

Dugdale and Goering (1967) suggest that one means of assessing the rate of phytoplankton production in an area, is by looking at the available forms of nitrogen for phytoplankton growth. These researchers proposed two forms of nitrogen available for uptake, 1) newly incorporated nitrogen as NO_3-N or N_2, and 2) recycled nitrogen in the form of NH_4^+-N or dissolved organic-N. Thus, two forms of primary production can be determined based on "regenerated nitrogen" (ammonium) or "newly available nitrogen" (NO_3-N or N_2-N), with the rate of export influencing which form prevails (Dugdale and Goering, 1967). This procedure also enables determination of the degree of autochthonous or allochthonous production within an area. However, as stated above, ammonium is not routinely analysed because of complications with the reliable measurement of low levels.

Selection of sample sites within estuaries or embayments requires consideration of a number of factors, including the number of samples which can feasibly be collected and

analysed. As mentioned previously, such systems exhibit spatial and temporal variability. For sampling along the axis of an estuary, Jassby et al. (1997) recommend selection of as many equally spaced stations as possible, with the selection of appropriate number and location of stations based on the variance obtained. Other researchers have used a single fixed station with frequent sampling over tidal cycles, generally monthly (e.g. Berg and Newell, 1986; Baird et al., 1987) or several fixed stations with sampling at high and low tide (e.g. Carver and Mallet, 1990). Another approach adopted is sampling of fixed stations located within channels at narrow points along estuaries, not always equidistant and generally extending from the upper region to the mouth, or beyond, with sampling conducted monthly (e.g. Hickman et al., 1991; Ball et al., 1997). Tett (1987) however, recommends that sampling should be conducted at the same state of the tide. Sampling at the time of low water has been assumed to estimate the tidal mean of parameters measured (Goulletquer and Bacher, 1988). The latter approaches were adopted in this study, with sample sites located in the upper regions beyond the oyster farm areas and extending out to marine sites.

This study presents data collected over a 13 month period from three oyster farming areas of different biotypes (marine, estuarine and intermediate). Variations in nutrients, phytoplankton biomass, seston quantity and quality, temperature, salinity and turbidity (secchi depth) were measured.

2.2 Materials and methods

2.2.1 Study areas

Three oyster farming areas in Southern Tasmania representative of marine (Pipeclay Lagoon), estuarine (Little Swanport) and intermediate (Pitt Water) systems were studied. The location of Pipeclay Lagoon, Little Swanport and Pitt Water are shown in Fig. 2.1. Sample sites were located from a marine site (largely outside the influence of the estuary/embayment) to the upper reaches beyond where oyster farms are located, with the exception of Pipeclay Lagoon which is an embayment and the uppermost site was located to capture the flow of water from the southern region of the lagoon. Sample sites were selected along the estuaries/embayments at the point of narrow constrictions or drainage channels. The rationale behind this was that water samples collected would be representative of the water flowing from the preceding section or segment.

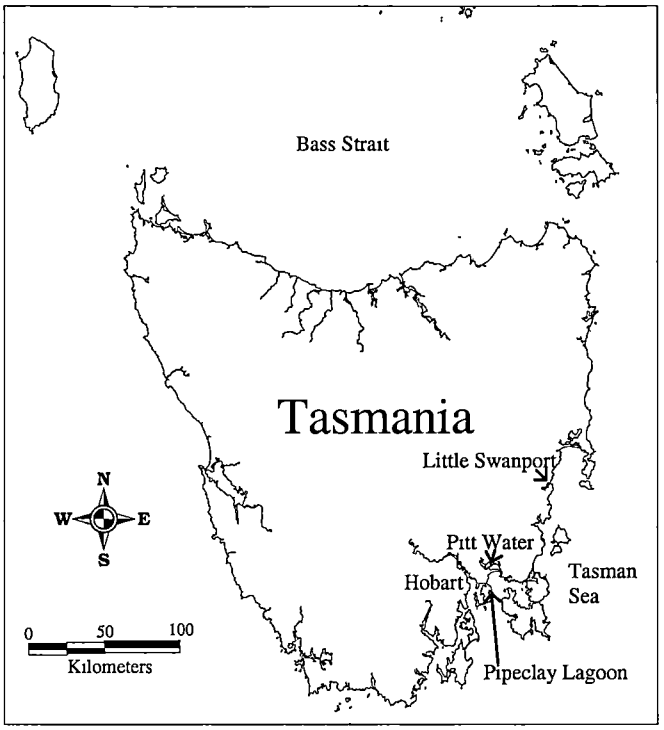

Fig. 2.1 Tasmania showing location of Little Swanport, Pitt Water and Pipeclay Lagoon.

The location of sample sites used in this study are similar to those used in the Fisheries Research and Development Corporation (FRDC) funded study titled "Predictive modelling of carrying capacities of oyster (*Crassostrea gigas*) farming areas in Tasmania" (Crawford et al., 1996). Additional sites were located within each of the areas, however the use of similar sites has enabled comparison with, and utilisation of, the historical data collected during the FRDC study.

2.2.2 Description and history of each of the study areas

2.2.2.1 Pitt Water

The Coal River drains into Pitt Water estuary, a region located approximately 30 minutes drive east of Hobart. The Coal River valley catchment is 630 km^2 (Anon, 1998) with an annual average rainfall of 500 - 600 mm and is prone to drought conditions. Pitt Water estuary (from the inlet entrance to the upper tidal reach, below Richmond) is approximately 21 km in length.

The Pitt Water estuary is complex and considerable man-made changes have altered the hydrology of this estuary. The first significant alteration occurred with the construction

of causeways from Tiger Head to Frogmore Peninsula (1st causeway) and between Frogmore Peninsula and Sorell (2nd causeway) (Fig. 2.2) in the early 1870s (Prestedge, 1995). The first causeway, which is approximately 1.5 km long, was constructed of rock fill with a wooden bridge of approximately 0.5 km length constructed near the eastern end. The second causeway, which is of similar length, had a narrow wooden bridge constructed approximately midway (Prestedge, 1995). During the 1950s the causeways were upgraded and the wooden bridge on the first causeway was replaced with a concrete bridge and the bridge across the second causeway was replaced with two concrete culverts forming Orielton Lagoon. Restricted tidal flows and inputs from the Midway Point sewage treatment plant (STP) into Orielton Lagoon instigated a remediation program in 1993 following complaints of odours and nuisance cyanobacterial blooms, predominantly *Nodularia spumigena* (Kinhill, 1993). Remedial action at Orielton Lagoon has included the increased opening of the two culverts into Pitt Water to facilitate greater tidal flow and exchange. Improvements have also been made to the operation and waste disposal from the Midway Point STP.

The second major change followed the construction of the Craigbourne Dam in 1986 across the upper reaches of the Coal River south-east of Colebrook creating the South East Irrigation Scheme. This significantly changed agricultural practices in the Coal River Valley from traditional wheat and sheep farming to more intensive, irrigated crops. As a consequence of this dam, the river flow pattern has changed to low consistent flows in summer and reduced flows in winter, almost a reversal of the former flow pattern. Additionally, the dam has acted as a buffer, with reduced flood events now occurring in the lower reaches of the estuary. A weir was later constructed approximately 0.5 km below the historic weir at Richmond in 1992, which additionally captured water flow further reducing fresh water inputs downstream.

Important habitat for a number of unique flora and fauna have been recognised within Pitt Water and are of such significance that part of the area was listed as a Ramsar Site Wetland Reserve in 1983 (DPIWE, 1999). Extensive areas of salt marsh are found in the upper reaches of the estuary, north of Lands End and Horatio Point. There is considerable silting of the river in this region, generally from catchment run-off, but also a significant amount occurs from deposition of wind blown soil from surrounding farmlands during dry periods and sparse pasture/crop cover (pers. obs.). Historically, flooding of the river regulated the build up of silt by flushing and dispersal to the lower

reaches. Such flood events are now rare and only occur following periods of prolonged and heavy rainfall.

The region of the estuary seaward of the first causeway is referred to as Lower Pitt Water, and the region above is known as Upper Pitt Water. Generally the substrate type and sediments between these two regions differ: Lower Pitt Water has more coarse sediments (medium-fine sand) and rippled/undulating relief indicative of stronger current flows, while Upper Pitt Water generally has much finer sediments with greater silt/clay content and a more even substrate relief suggestive of lower current flows (Mitchell et al., 1998). The bed formations of Lower Pitt Water, in particular, were found to have changed little from the features described by Harris (1968) in his study of sedimentology of Pitt Water (Mitchell et al., 1998). Extensive beds of seagrass formerly existed in the estuary, though massive loss has occurred since the late 1940s (Rees, 1993). Rees (1993) reported declines from 1276 ha in 1948 to 585 ha in 1969 to 75 ha in 1990, a 94% decrease over 45 years. Sparse beds of *Zostera muelleri* exist in Upper Pitt Water (pers. obs) with variable cover of seagrass (*Heterozostera tasmanica* and *Zostera muelleri*) observed in Lower Pitt Water. Geoff Prestedge (pers. comm.) has noted small recovery of beds in Lower Pitt Water in recent years. His diary provides a valuable historical record of changes in the fauna and flora of the lower estuary region from observations made since 1956 (Prestedge, 1995).

The first oyster leases were established in Upper Pitt Water in the early 1980s in the region along the western shore north of the spit and Barilla Bay (Fig. 2.2). Currently there are 7 intertidal marine farms ranging from 10 ha to 20 ha occupying a total area of 108.19 ha (DPIWE, 2000). All leases culture Pacific oysters (*Crassostrea gigas*), though some attempts have been made at growing native/flat oysters (*Ostrea angasi*).

2.2.2.2 Pipeclay Lagoon

Pipeclay Lagoon is located on the south-western side of Fredrick Henry Bay, South Arm Peninsula and is approximately a 40 minute drive from Hobart. It is a relatively shallow body of water with tidal exchange of sea water from Frederick Henry Bay via a narrow opening (approximately 150 m width) at the southern end of Cremorne Beach (Fig. 2.3). The area of the lagoon is approximately 532 ha (Brown and Mitchell, 1991), with large areas of shallow water and extensive areas of sand flats exposed during low tide. Approximately, 86% of the lagoon area has water depths less than 2 m. The catchment

area is relatively small with minimal fresh water inputs to the lagoon. No permanent creeks flow directly into the lagoon. Freshwater inputs to the lagoon only occur after prolonged and heavy rainfall following saturation of the swamp/marsh land at the north (Rushy Lagoon) and southern ends of the lagoon.

The region has a moderate average rainfall and is frequently subject to drought conditions. Average annual rainfall recorded at Clifton Beach (located south of Pipeclay Lagoon) for the period 1982 to 1988 was 577 mm (Brown and Mitchell, 1991), with a relatively even distribution of rainfall throughout the year.

Pipeclay Lagoon is approximately 4 km in length (mid-water). The lagoon is shallow, with a deeper, relatively narrow channel, flowing from the mouth of the lagoon to a deep hole at the south end. Several gutters branch from this channel and pass through the main lease area. Dense beds of seagrass (*Heterozostera tasmanica*) occur in the lagoon, particularly in the region from the mouth to Bens Gutter, with small beds occurring elsewhere, generally in the channel or gutters (Mitchell and Macleod, 1998). These beds of seagrass appear to have increased within the previous 3-4 years (pers. obs.). Mitchell and Macleod (1998) found that the central region of the lagoon was characterised by a general substrate type of fine sand with varying amounts of shell debris and it was noted that shell debris, predominantly flat oyster (*Ostrea angasi*), was observed within the sub-surface at some sites, presumably former beds subsequently buried.

Pipeclay Lagoon is one of the pioneering areas of commercial cultivation of Pacific oysters. The first leases were granted in the early 1970s with additional leases granted in the late 1980s (DPIF, 1998). Currently there are seven operational marinc farms culturing Pacific oysters. Farm sizes range from 2.52 to 13.03 ha, with a total area of 48.25 ha (DPIF, 1998). All farms are within the eastern region of the lagoon (Fig. 2.3) and occupy an area of approximately 9% of the lagoon. A commercial oyster nursery was established in 1985 adjacent to the lagoon (near the mouth) which has expanded considerably and is one of the major suppliers of oyster seed to industry.

2.2.2.3 Little Swanport

Little Swanport is situated on the East Coast of Tasmania and is approximately a 1.5 hour drive from Hobart. The east coast of Tasmania has a moderately low average rainfall and at times is subject to drought conditions. Mean average rainfall recorded at

Ravensdale (near Little Swanport) for the period 1942 to 1987 was 658 mm (Mitchell, 1988). The Little Swanport estuary is approximately 7.5 km in length with tidal exchange of marine water from Great Oyster Bay through a single narrow opening, approximately 150 m in width, at Limekiln Point. Flow of marine water into the estuary is through a narrow deep channel on the northern side. There is a shallow sandbar on the southern side which is submerged at high tide.

The Little Swanport River is the main fresh water input into the estuary with contributions from Ravensdale Rivulet, White Hut Creek, Boomer Creek and Brushy Creek (Mitchell, 1988). The catchment area for this river system is 597 km^2 to the west of the estuary rising up to the Tiers which form the lower central plateau region of Tasmania (Mitchell, 1988). Flooding of the various water courses that drain into this estuary occurs following periods of prolonged and heavy rainfall. On these occasions the salinity of the estuary water is considerably depressed, with the sandbar creating a barrier to the exchange and flushing of the estuary. Salinity within the estuary can be variable during these times, depending upon the prevailing wind conditions which influence the degree of water mixing and fresh water flow.

Most of the water courses which drain into the estuary pass through bush or forested area. Marginal areas have been cleared for pasture, predominantly for sheep, with superphosphate applied for pasture improvement. The upper region of the estuary has extensive areas of mudflats exposed at low tide with dense reed and ricegrass (*Spartinia anglica*). Within the mid to lower region of the estuary are extensive beds of eelgrass (*Zostera muelleri*). Beds exist adjacent to and within existing operational marine farms in this region of the estuary. These beds have been little affected, or disturbed, from the oyster farm activities due to physical means, such as outboard motor propellers, being walked upon, or oyster cultivation with, the exception of areas directly under racks, attributed to shading (pers. obs.). Expansion of *Zostera muelleri* within the estuary has occurred. Rees (1993) reported an increase within the region from Plentiful Point to the mouth from 65 ha in 1950 to approximately 90 ha in 1990, an increase of 38%. Extensive areas of cover, not mapped by Rees (1993), have been observed further up the estuary (pers. obs.).

There are currently three operational marine farms within the estuary (Fig. 2.4) which have been in operation since the 1980s. Farm sizes are 52.7, 10 and 16.8 ha, and occupy a total area of 79.5 ha (DPIF, 1997).

2.2.3 Sample collection

Seven sample sites were located along the Pitt Water estuary from the Marine site (site 1), located near Spectacle Head at Dodges Ferry on the inside of Speck Island, to beyond the lease areas in the upper estuary reaches (Top End - site 7) (Fig. 2.2). Total length of the estuary sampled was approximately 17.5 km.

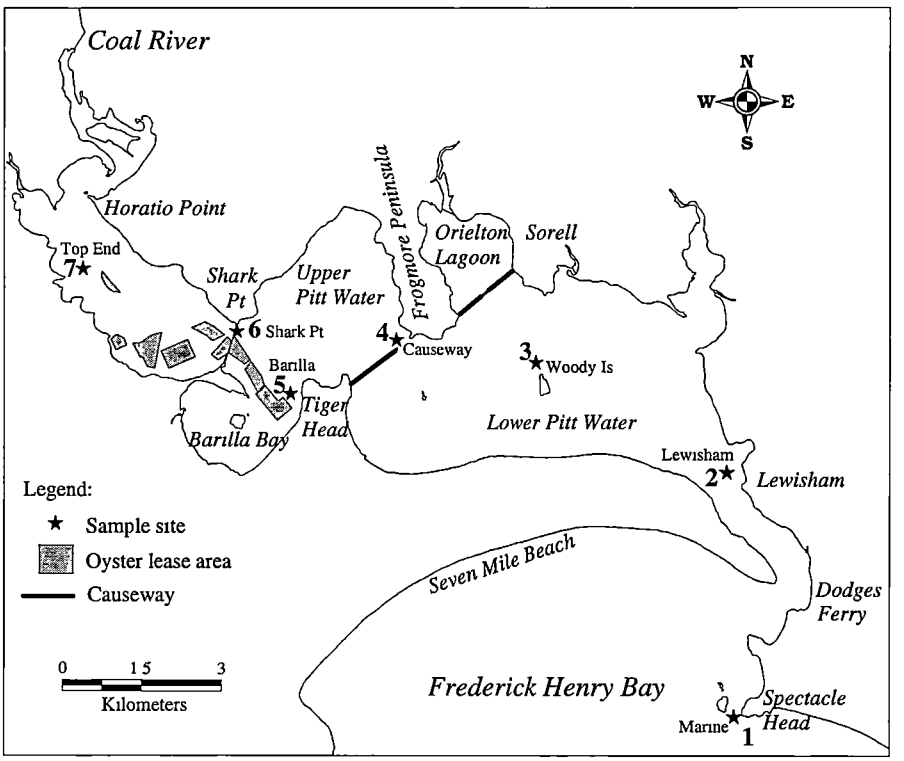

Fig. 2.2 Location of sample sites in Pitt Water estuary (source of oyster lease locations DPIWE, Marine Farming Branch).

Water samples were collected from five sites in Pipeclay Lagoon, a Marine site (site 1) located outside the influence of the lagoon and four sites within the lagoon (Fig. 2.3). Site 3 was selected to collect water draining from Bens Gutter, a region which incorporates much of the lease areas. Nemo (Site 5) was located in the channel which drains from the deep hole in the southern region of the lagoon.

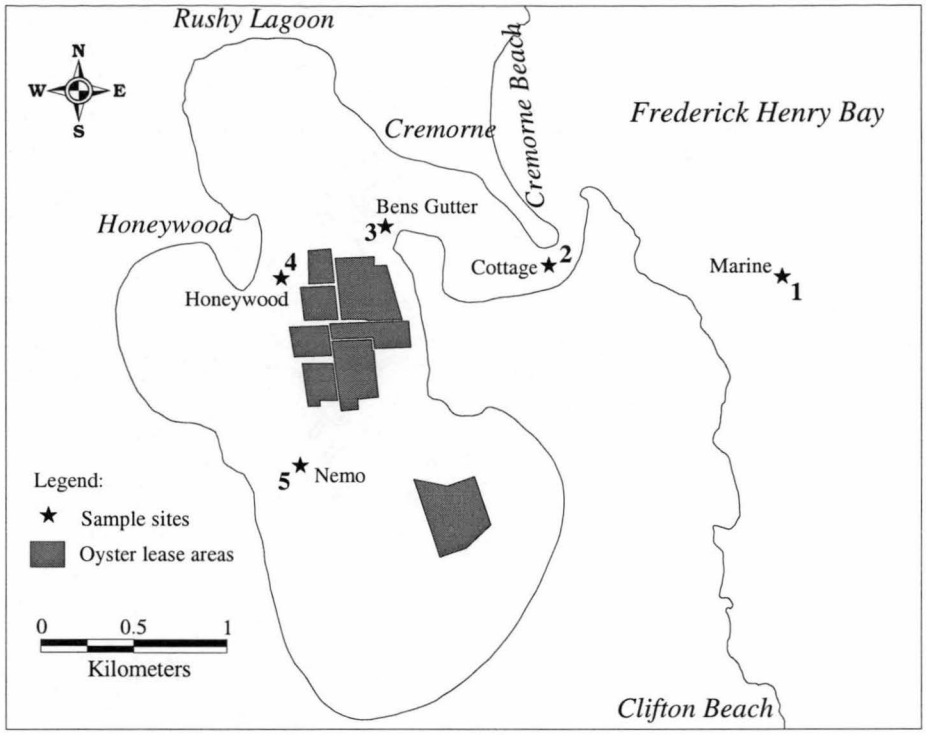

Fig. 2.3 Location of sample sites in Pipeclay Lagoon (source of oyster lease locations DPIWE, Marine Farming Branch).

Little Swanport initially had five sample sites, but after the first sampling session it was felt an additional sample site should be located to capture the bifurcated flow of water out of the estuary from the south eastern side (Shack - site 3) and flow from the eastern side of Ram Island (Jacks Island - site 4). Thus, an additional station was located in the entrance channel (Limekiln - site 2) which was sampled thereafter (Fig. 2.4). Approximate distance between Limekiln and the upper estuary site (Dyke - site 6) was 6.5 km.

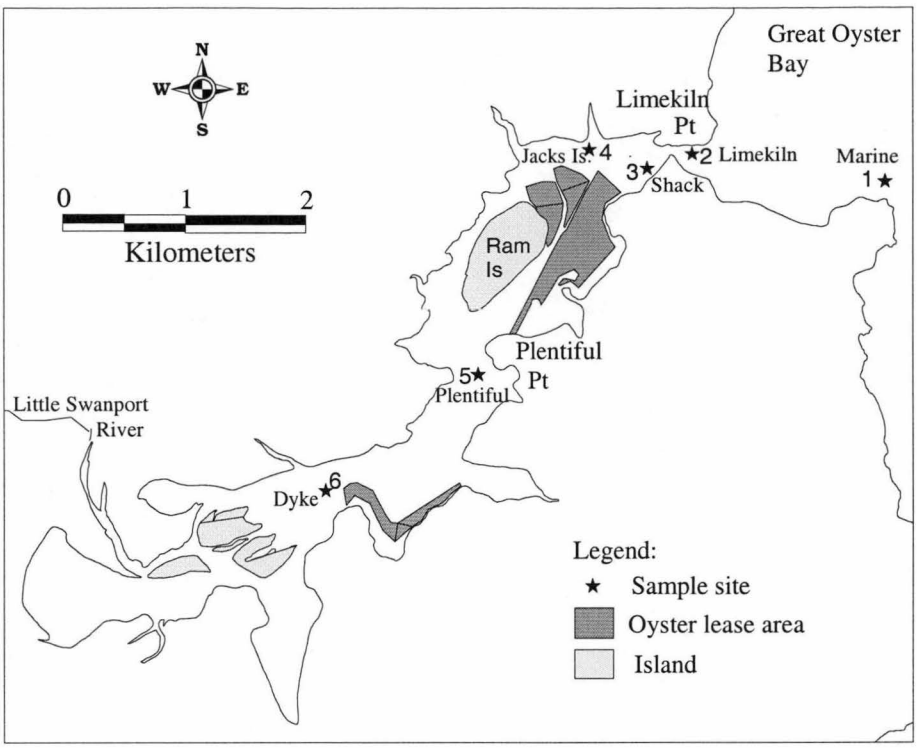

Fig. 2.4 Location of sample sites in Little Swanport estuary (source of oyster lease locations DPIWE, Marine Farming Branch).

Two water samples were collected at each site using time-integrated sample bottles. These bottles sampled approximately 8 litres of water over a 50 minute period. More detailed description of the design and evaluation of the time-integrated water sample bottles is provided in Appendix 1.1. Time-integrated sample bottles were selected over traditional "grab" samples (where instantaneous samples are collected at one point in time) because more representative water samples would be collected (Fabris et al., 1982) reducing any effect of "patchiness' in water column variables (i.e. phytoplankton, seston or nutrients). On deployment, each bottle was attached to an anchored surface float with the intake tube set approximately 1 m below the surface. All bottles were set out at fixed locations along the estuary in quick succession.

Water samples were collected approximately monthly over 13 months from each site on an outgoing ebb tide prior to low (slack) water. Times of low tide were obtained from the Tasmanian Tide Tables (issued by the Tasmanian Port Authorities). Time of deployment, temperature, salinity and secchi disk depth were recorded at each station, and details of prevailing weather conditions (cloud cover, approximation of wind speed and direction, and precipitation) were noted.

All samples were processed within 12 h of collection, generally within 3 h. Each sample was thoroughly mixed prior to decanting sub-samples for nutrient, chlorophyll and particulate matter measurements. Duplicate 10 ml samples were collected in sterile tubes and frozen (-20^0 C) for later nutrient analyses. 0.2 - 0.8 L was used for chlorophyll a determinations, depending on the quantity of particulate matter in the water sample. The sample was filtered using reduced vacuum pressure through 47 mm Whatman GF/F glass fibre filters (0.8 μm nominal pore size), the filters carefully folded with concentrate on the inside surface and frozen (-20^0 C) in labelled sterile tubes. For particulate matter (seston), between 0.2 and 1.0 L (generally 0.5 L) of sample was filtered, depending on the particle loading (see Appendix 1.1) through pre-combusted (480^0 C for 16 h) and pre-weighed 47 mm Whatman GF/F filters using low vacuum pressure.

2.2.4 Pilot study

Assessment was made of the changes in nutrient, chlorophyll a, temperature and salinity over time at two stations in Pitt Water: site 4 (North Causeway) and site 6 (Shark Point). This investigation was conducted in collaboration with Dr Christine Crawford as part of the FRDC project, but also as a pilot for this study. Results of the intensive sampling study are detailed in the final report of that project (Crawford et al., 1996). A copy of the results is provided in Appendix 1.2. Briefly, replicate (n=3) water samples were collected using time-integrated water sample bottles every day on the ebb tide (prior to slack water) for 1 week, followed by once a week for four weeks. The chlorophyll a data used in this assessment has been performed on chlorophyll a values as calculated using the formula adopted in this study and not the formulae used by Crawford et al. (1996).

2.2.5 Phytoplankton biomass

Phytoplankton biomass was estimated from chlorophyll a determinations using a modified APHA (1985) Standard Method 1002G. Frozen filters were torn into small pieces using forceps and placed in a glass screw cap tube. A known volume of 90% acetone was added (11-12 mls), and the sample was sonicated for approximately 15–30 seconds to disrupt the filter fibres and phytoplankton cells before centrifuging at high speed (approx. 11 x 10^3 rpm). The absorbance of the supernatant extracted was read at

663 nm and 750 nm in a 4 cm path length cuvette using a Pye Unicam Spectrophotometer. The extract was then acidified using dilute hydrochloric acid (50/50 HCl:distilled H_20), gently mixed and the absorbance at the same wavelengths measured. The spectrophotometer was zeroed at 663 nm using 90% acetone in a 4 cm cuvette. The blank for acidified extracts was conducted using 90% acetone and the same amount of dilute acid as was added to sample extracts, without alteration of the blank wavelength setting. A maximum of 8 samples were analysed at the one time, with the extraction process and spectrophotometer readings carried out in subdued light, and the sample extracts stored on ice to minimise degradation of chlorophylls. Each absorbance reading was corrected for the equivalent blank, that is the pre- and post-acidification 663 nm readings were subtracted from the equivalent pre- and post-acidification 750 nm readings. Chlorophyll a and chlorophyll a degradation products (phaeopigments) were calculated using the formula from Tett (1987):

$$\text{Chlorophyll a } (\mu g\ L^{-1}) = \frac{27.7\ (663_b - 663_a)\ V_1}{(V_2)\ L}$$

$$\text{Phaeophytin a } (\mu g\ L^{-1}) = \frac{27.7\ (1.7\ (663_a - 663_b))\ V_1}{(V_2)\ L}$$

where V_1 = volume of solvent sample extract (ml)

V_2 = volume of water sample filtered (L)

L = length of cuvette (cm)

663_a = absorbance after acidification (less acidified 750 nm reading)

663_b = absorbance before acidification (less 750 nm reading)

2.2.6 Nutrients

Nutrient analyses were conducted using a four channel Skalar® Segmented Flow Analyser. Modified Skalar (1993) methods were used from advice given by researchers at the Australian Institute of Marine Science (AIMS) laboratories (Townsville) and the Queensland Government Chemical Laboratories (Brisbane). Low level nutrient standards within the analytical range of samples analysed were used for determination of sample concentrations. On occasions, samples were diluted to within the range of the standards routinely used.

Nitrate + nitrite (generally referred to as NOX) was determined by the cadmium reduction method, where the sample is passed through a column containing granulated

copper-cadmium to reduce nitrate to nitrite (Collos et al., 1992). The resulting reduced nitrite (+ original nitrite content) was determined with sulphanilamide coupled with α-naphthylethylenediamine dihydrochloride (NED) to form a reddish coloured azo dye measured at 540 nm (Skalar 1993).

Nitrite (NO_2-N) was determined using the preferred option of the NOX channel with the cadmium column switched off with a similar resultant reaction product. Nitrate (NO_3-N) content could be determined from the NOX result less the nitrite value.

Phosphorus (ortho-phosphate PO_4-P) was determined by the reaction of ammonium molybdate and potassium antimony tartate in an acidic medium forming an antimony-phospho-molybdate complex, which was reduced to a blue coloured compound by ascorbic acid and measured at 880 nm (Skalar, 1993).

Silicon (SiO_4-Si), often expressed as silicate, was determined from the reaction of acidified sample mixed with ammonium molybdate solution to produce molybdosilicic acid. This acid was reduced with ascorbic acid to produce a blue dye measured at 810 nm (Skalar, 1993).

All standards were freshly prepared on the day of nutrient analyses, and analytical sample runs consisted of 10 samples/standard. On occasion duplicate samples and/or standards were run as a method and performance calibration check. This procedure was also conducted frequently as a check on the cadmium column efficiency, and when noted to be declining, the column was reconditioned (i.e. repacked) and trialed before continuing with sample analyses (Garside, 1993).

All samples, frozen after collection, were thawed at room temperature on the day of analysis. Not all analyses could be performed within 1 week of collection. Preliminary investigations conducted by the Qld. Govt. Chem. Labs. suggest that deterioration (loss) of nitrite occurs in samples stored frozen for longer than 1 wk (pers. comm. Dan Wruck, Qld. Govt. Chem. Labs., Brisbane). As a result of this, while some samples were analysed within a few days of collection, only NOX values have been used. Additionally, long thaw times were allowed prior to silicon determinations (minimum 2 - 4 hrs) as this increased the recovery of the 'reactive' fraction (pers. comm. Ruth Erikson, CSIRO, Hobart). Unfortunately it was not possible to conduct ammonium analyses on samples collected.

2.2.7 Particulate matter/seston quantity and quality determinations

Assessment was made prior to sample collection to determine an appropriate method for total particulate matter (TPM), particulate organic matter (POM) and particulate inorganic matter (PIM) concentrations. A method was developed using low volumes, low vacuum pressure and rinsing filters with ammonium formate. An apparatus was designed and constructed for this purpose, to enable regulation of vacuum pressure and a one litre glass vessel made to enable single decanting of water samples for filtration.

For determination of TPM, POM and PIM, water samples were filtered through pre-ashed (480^0 C for 16 h) and pre-weighed 47 mm Whatman glass fibre filters (GF/F) using low vacuum pressure (20-35 psi). The vacuum was disconnected and the filters rinsed with 0.9% ammonium formate to remove salts. This was then filtered through the filters by re-connecting the vacuum. The filters with particulate matter were removed and placed in labelled petri dishes then into a cool oven (~ 20^0 C) initially to prevent the filters sticking to the base of the dishes. The filters were dried (24-36 h) at 65^0 C and weighed to 0.01 mg accuracy using a Mettler Toledo micro-balance. TPM was determined from the dry weight of the filter plus sample minus the weight of the filter paper.

The filters were placed on small aluminium dishes and ashed at 480^0 C for 4 h in a pre-heated furnace. The filters were allowed to cool in a desiccator before re-weighing to 0.01 mg. POM was calculated from the difference in the oven dry and ashed weight, and PIM determined from the weight remaining (less initial filter weight) after ashing.

2.2.8 Field measurements

Temperature and salinity was measured at 1 m depth sub-surface using a WTW LF196 Temperature/Salinity meter. Secchi depth was measured using a 24 cm black and white weighted circular secchi disk with graduated line.

2.3 Results

Time series plots for the 13 month period for each of the parameters measured at sites in each area are shown (Fig. 2.5 to Fig. 2.16). Strictly lines should not be drawn to connect points between successive sampling periods, given the variability which can occur over shorter time scales than the sampling period adopted during this study. However, for

convenience lines have been drawn between points to show the general trend over the sampling times. The raw data for temperature, salinity, rainfall, secchi depth, chlorophyll a, nutrients and seston at each site are given in Appendix 1.3.

There are only a few missing data points. The marine sites of Pipeclay Lagoon and Little Swanport were not sampled in January 1996 because of rough seas. No samples were collected in July 1995. On occasion some sample bottles deployed at shallow sites drifted, or were blown, into shallower water resulting in sediment uptake causing aberrant results. These were omitted from the data analyses.

2.3.1 Pilot study

Detailed plots of this study are provided in Crawford et al. (1996). Temperature variation over the 4 week period was approximately 3^0 C (13.5 - 16.5^{0}C) with the upper site (Shark Point) showing the higher values. Salinity showed a similar trend, with the variation during this time approximately 1.2 ‰ (33.3 - 34.5 ‰). Chlorophyll a and silicon showed daily and weekly variation. NOX levels were low and marginally above the detection level. Mean phosphate levels showed some variation and Shark Point generally had the higher concentrations.

The data were analysed to determine the range of variability in each parameter over the month. This was based on the mean values (n=3) measured over the 7 day and weekly periods at each of the two sites (Table 2.1). This analysis provides some means of assessing the degree of variability which may occur when sampling is conducted at only one time during a month period. However, it must be acknowledged that this was based on results for a single month period.

Table 2.1 Mean minimum, maximum and average values of water column parameters measured at two sites in a month period at Pitt Water.

Site	Variable	Minimum	Maximum	Average
Shark Point	Temperature (C)	13.7	16.3	15.3
	Salinity (ppt)	33.7	34.5	33.9
	NOX–N (µg/L)	0.1	1.7	1.0
	PO_4–P (µg/L)	10.5	11.5	10.9
	SiO_4–Si (µg/L)	161.3	248.7	207.0
	Chlorophyll a (µg/L)	0.061	1.225	0.825
North Causeway	Temperature (C)	13.5	15.4	14.7
	Salinity (ppt)	33.3	34.2	33.7
	NOX–N (µg/L)	0.7	2.3	1.3
	PO_4–P (µg/L)	8.9	12.6	9.8
	SiO_4–Si (µg/L)	130.7	209.0	165.0
	Chlorophyll a (µg/L)	0.441	1.593	0.888

2.3.2 Pitt Water

2.3.2.1 Temperature, salinity and *secchi depth*

Temperature showed a seasonal trend with June the coldest month (Fig. 2.5a). The shallowest site, located at Barilla Bay (site 5) showed the most extremes of temperature in the colder and warmer months (4.4^0 C and 23.8^0 C, respectively). Generally, since sampling was conducted on an ebb tide (near slack water), the estuary sites showed lower temperatures than the marine site in the colder months and higher temperatures in the warmer months. This most likely was due to the influence of ambient air temperatures either heating, or cooling, the water flowing from the upper, shallower estuary regions.

Little variation in salinity was shown at the marine site, but the within estuary sites showed degrees of variability with the greatest change due to freshwater inflow following rainfall events (Fig. 2.5b). During the warmer periods the estuary sites showed elevated salinities (hypersaline conditions) due to evaporation. There were relatively heavy rains 12 days prior to sampling in April 95 (Fig. 2.5c), though no reduction in salinity within the estuary was shown with the exception of Lewisham which had marginally lower salinity compared to the other sites. This site recorded high NOX levels at this time, suggesting some localised freshwater input.

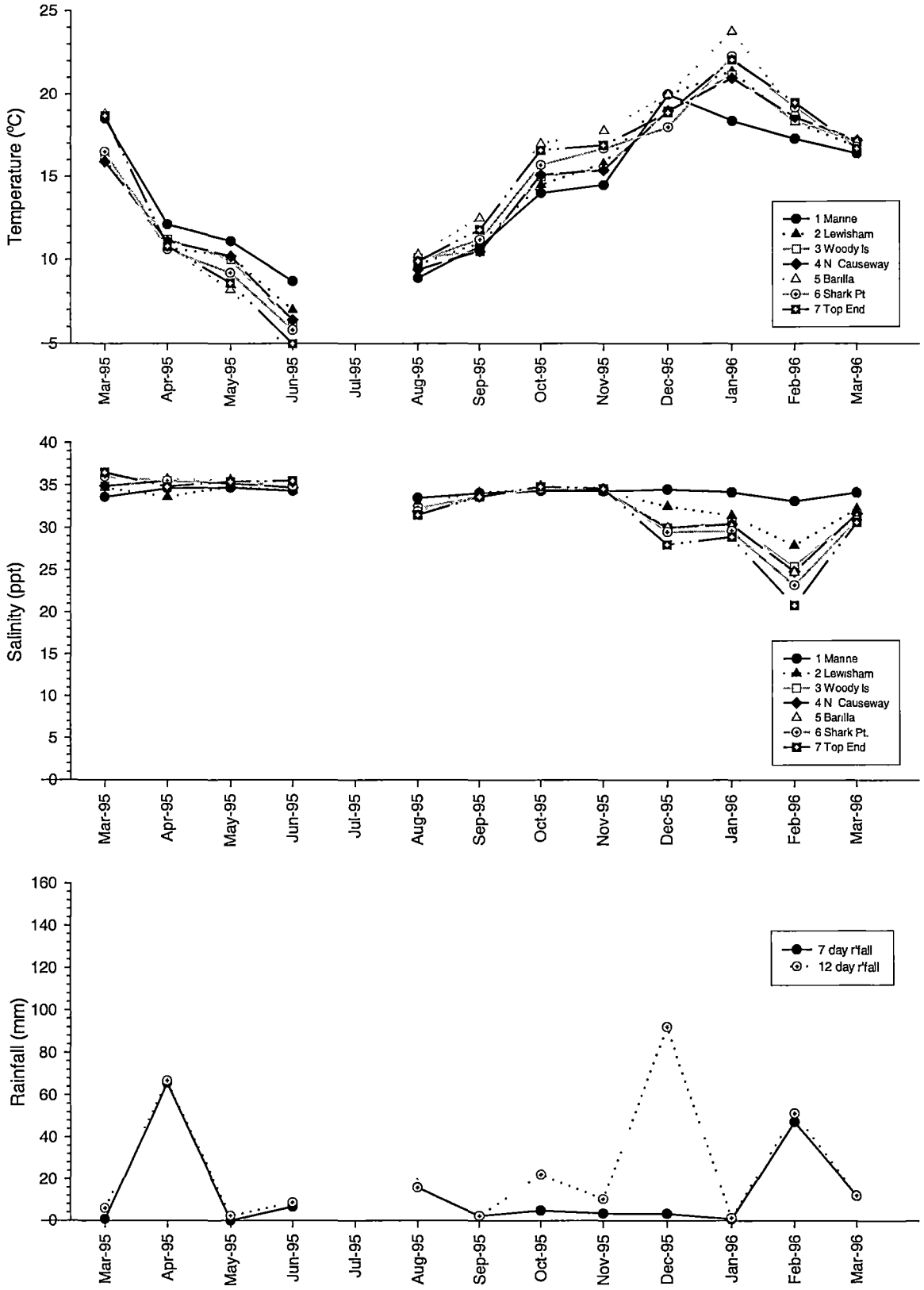

Fig. 2.5 a, b and c. Temperature, salinity, 7 and 12 day cumulative rainfall in Pitt Water.

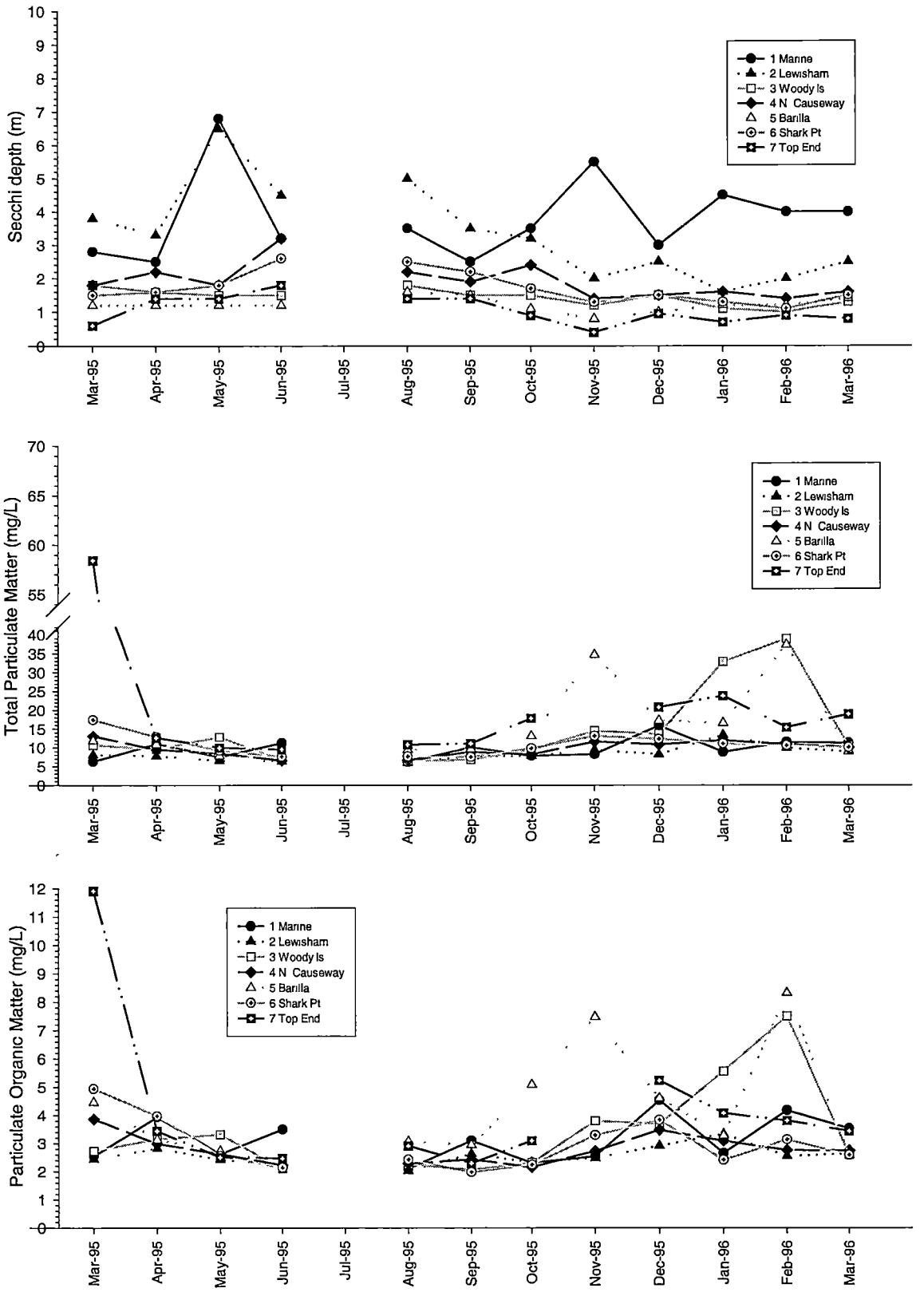

Fig. 2.6 a, b and c. Secchi depth, total particulate matter and particulate organic matter concentrations in Pitt Water.

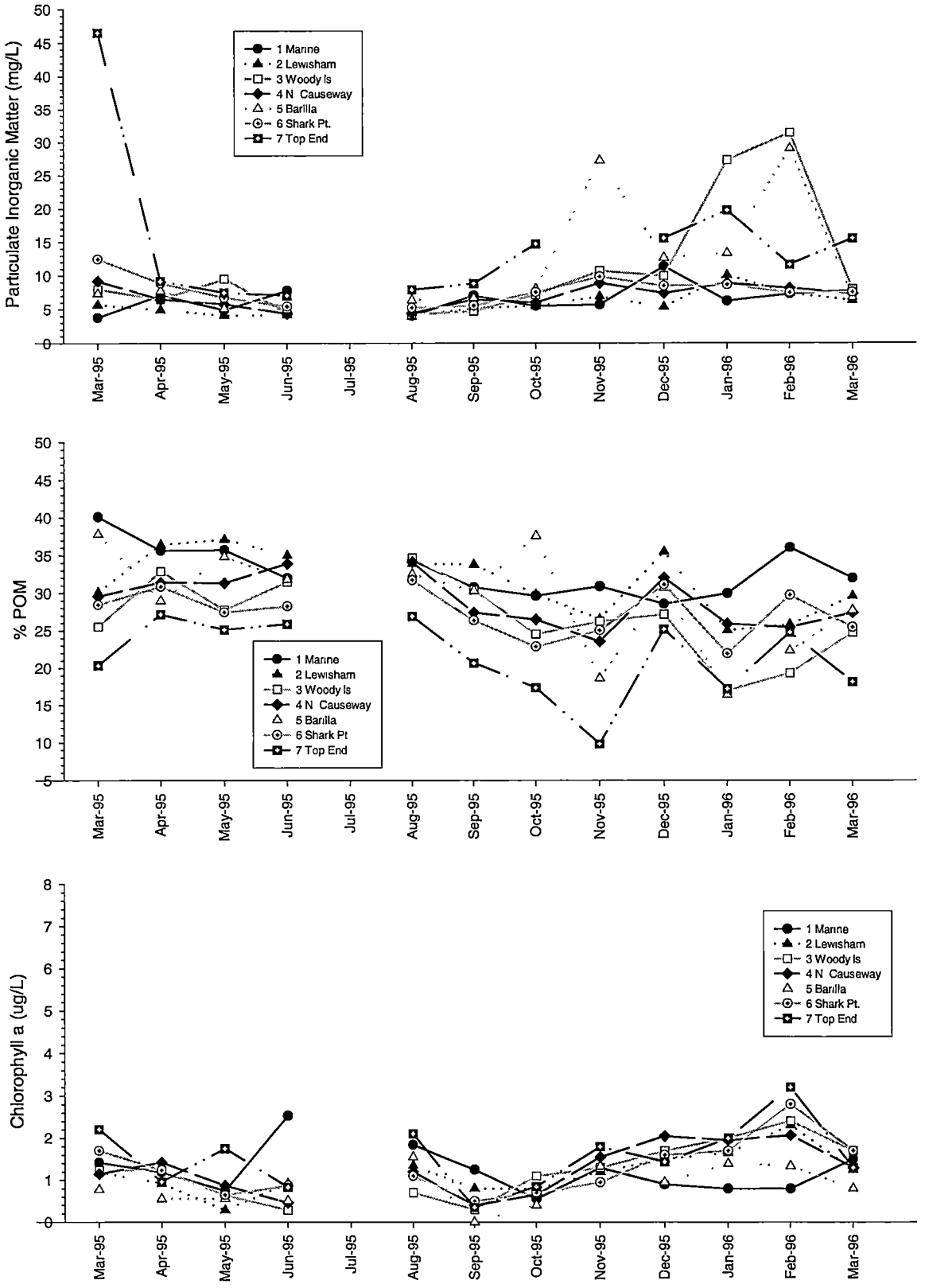

Fig. 2.7 a, b and c. Particulate inorganic matter, percentage particulate organic matter and chlorophyll a concentration in Pitt Water.

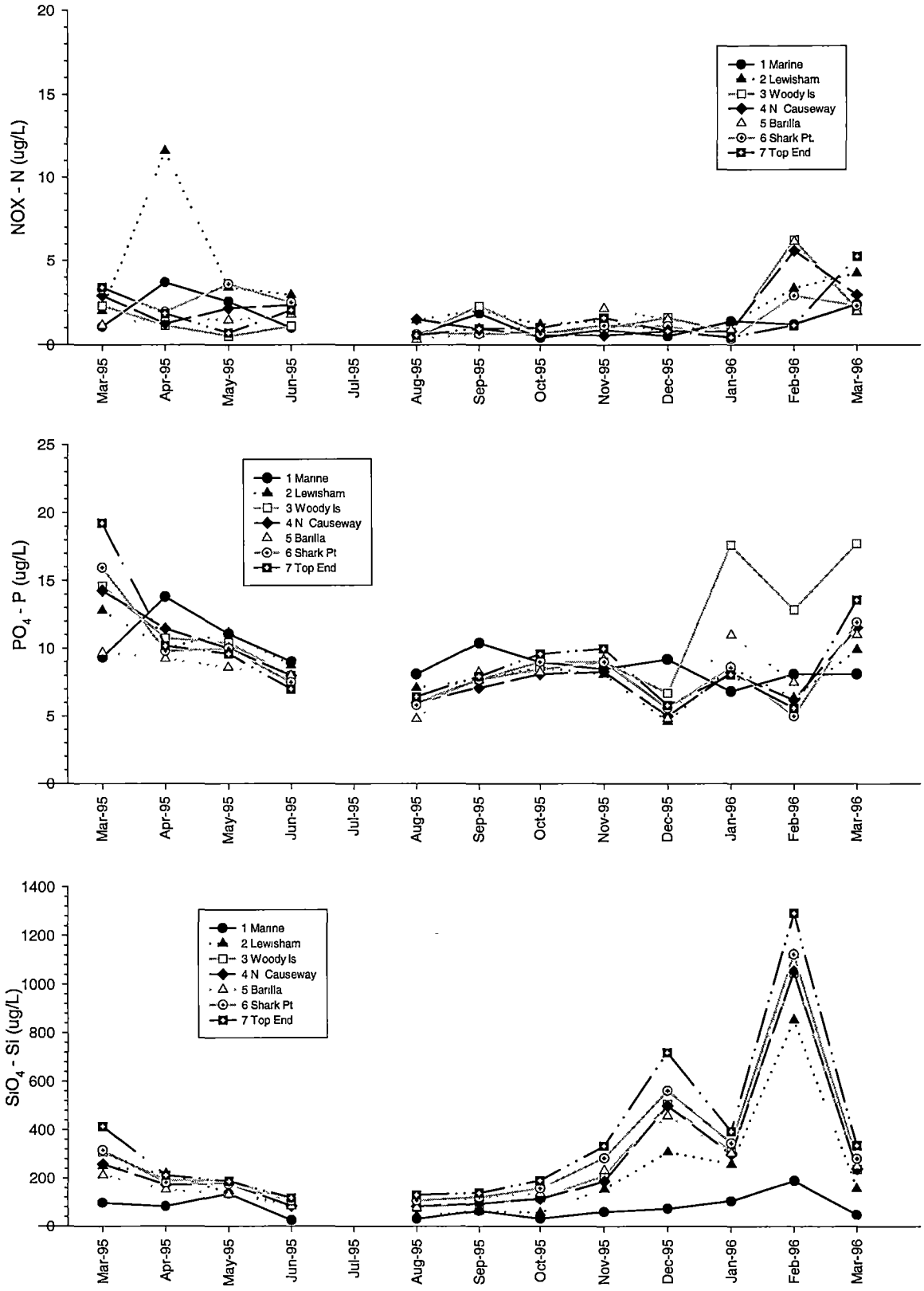

Fig. 2.8 a, b and c. NOX, PO_4-P and SiO_4-Si concentrations in Pitt Water.

Frequent rains during July and early August resulted in slight reductions in salinities within the estuary at time of sampling in August. Heavy rainfalls occurred in December resulting in depressed salinities from Woody Island to the Top End in particular, with slow recovery as shown with salinities recorded during January 96. Salinity at Lewisham was depressed to a lesser degree during this period. Further heavy rains during late January to mid-February 96 caused a significant reduction in salinities at all the estuary sites, with the upper sites recording the lowest salinity. Salinities at these sites were still relatively low, as compared to the marine site, through to March 96.

Secchi depth (Fig. 2.6 a) was variable along the estuary, with the disk still visible on the bottom at the shallow sites of Woody Island and Barilla Bay, though often the bottom was not discernible. However, the uppermost site (Top End) which was in approximately 2.5 m of water, secchi depths were relatively shallow and less than the water depth on most occasions. The water at this site was always observed to be turbid due to suspension of silt/clay particles. A similar observation was noted at Shark Point, where despite the water depth being approximately 7-8 m, shallow secchi depths were recorded with little variation shown. Similarly, the site located north of the causeway (site 4) also showed little variation, with turbid water observed at all times. This site experiences strong current flows due to the narrow constriction at the causeway bridge.

Lewisham, the deepest site (approximately 10-11 m depth), showed the greatest variability in secchi depth, which also followed the seasonal trend of temperature, with greater depths in the colder months and shallower depths in the warmer months. Secchi depths at the Marine site unfortunately were not representative. On occasions the exact site could not be reached due to sea conditions (large swell or breaking waves), therefore sampling was conducted approximately 100-200 m short of the site and hence in shallower water. Generally water depth at these times was approximately 4 m and the bottom clearly visible.

2.3.2.2 Seston quality and *quantity*

The highest TPM concentrations were consistently recorded at the three sites north of the causeway (Barilla Bay, Shark Point and Top End) (Fig. 2.6 b). At times TPM levels measured at Woody Island and Barilla Bay were relatively high, most likely due to wind driven resuspension of sediments at these shallow sites.

At the time of sampling, prevailing weather conditions were noted (cloud cover, estimate of wind speed and direction, precipitation). This information is of interest when interpreting seasonal differences; near calm conditions were experienced at the time of sampling in the autumn and winter months of 1995 (April - August), whereas variable wind strengths were encountered on most other occasions. Additionally freshwater inputs, particularly during December to January, could also account for increased suspended matter in the water column.

POM was within the range of 2 - 5 mg L^{-1} with the higher levels frequently recorded at the Upper Pitt Water sites (Fig. 2.6 c). On occasion, high levels were measured at the Top End, Woody Island and Barilla Bay sites. PIM levels were in the range 5-15 mg L^{-1} (Fig. 2.7 a) and showed a similar trend to POM. %POM was within the range of 26-37% for most sites (Fig. 2.7 b) with the exception of the Top End site which had lower values within the range 10-27%, reflective of the fine sediment suspension in the water column at this site. A trend of reducing %POM along the estuary from the Marine site was shown. Marginal increase in %POM was shown on occasion following rainfall events and hence organic matter input from catchment run-off. Low %POM was measured at the Top End site in November 95, a time when strong winds (20-30 knots) occurred during sampling.

2.3.2.3 Phytoplankton biomass

Chlorophyll a concentrations were low, less than 3.5 μg L^{-1} (Fig. 2.7 c). A seasonal trend is apparent with marginal decline in late-summer to autumn (March 95 to May 95), and an increase during the colder winter months (June 95 to August 95), particularly at the marine site. Levels were low in early spring (September 95), increasing from late spring through summer for the estuary sites. On occasion Top End had the highest readings. Higher chlorophyll a concentrations occurred at the estuary sites during December 95 to February 96 coinciding with depressed salinities, and were greater than the marine site during this same time.

2.3.2.4 Nutrients

NOX was low (Fig. 2.8 a), with the exception of Lewisham in April 95 as described in the salinity section above. NOX concentrations never fell below non-detectable levels. The trend of reduced NOX with elevated chlorophyll a was apparent. However, higher

NOX concentrations were measured in February 96 concurrently with elevated chlorophyll a levels. This most likely was attributed to the freshwater inflow during December - January.

Ortho-phosphate concentrations were low (Fig. 2.8 b), but higher than the NOX concentrations. Comparing the time series plot with the chlorophyll a data seems to indicate a similar trend to that noted with NOX and suggests biological uptake. Higher levels were recorded when chlorophyll a was low and vice versa. Higher concentrations were measured at Woody Island during January-March 96, coinciding with the period of reduced salinities, though the other sites did not show this trend.

The highest levels of silicon were consistently recorded at the uppermost site of the estuary, Top End (Fig. 2.8 c), and reflect the elevated levels of suspended sediments observed at this site. Levels decreased seaward along the estuary towards the marine site. Little variability was shown in silicon concentrations at the marine site with reduction coinciding with an increase in chlorophyll a suggesting biological uptake (possibly by diatoms, though no phytoplankton net samples were collected and examined). Considerable increase in silicon concentrations followed freshwater inputs during December - March. Elevated levels were also recorded on occasions most likely due to resuspension of sediments when sampling in windy conditions.

2.3.3 Pipeclay Lagoon

2.3.3.1 Temperature, salinity and *secchi depth*

A similar seasonal trend to Pitt Water was shown with June the coldest month (Fig. 2.9 a). However, the minimum temperature was slightly higher (7.8^0 C) and maximum temperatures lower ($< 20^0$ C). The sites within the lagoon had similar temperatures during the sampling period. The marine site showed marginally higher temperatures in the cooler months and lower temperatures in the warmer months as compared to the within lagoon sites at similar time periods.

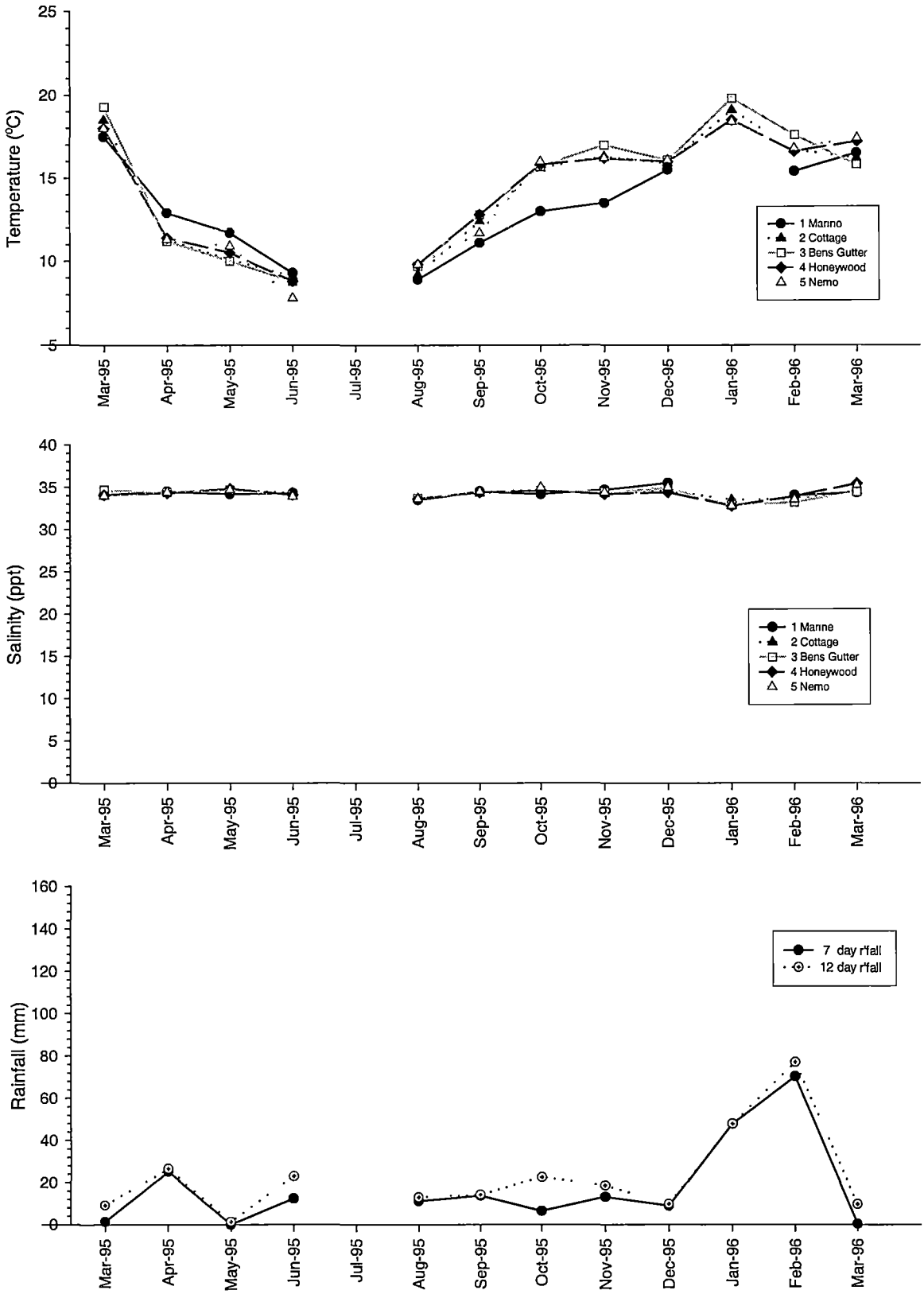

Fig. 2.9 a, b and c. Temperature, salinity, 7 and 12 day cumulative rainfall in Pipeclay Lagoon.

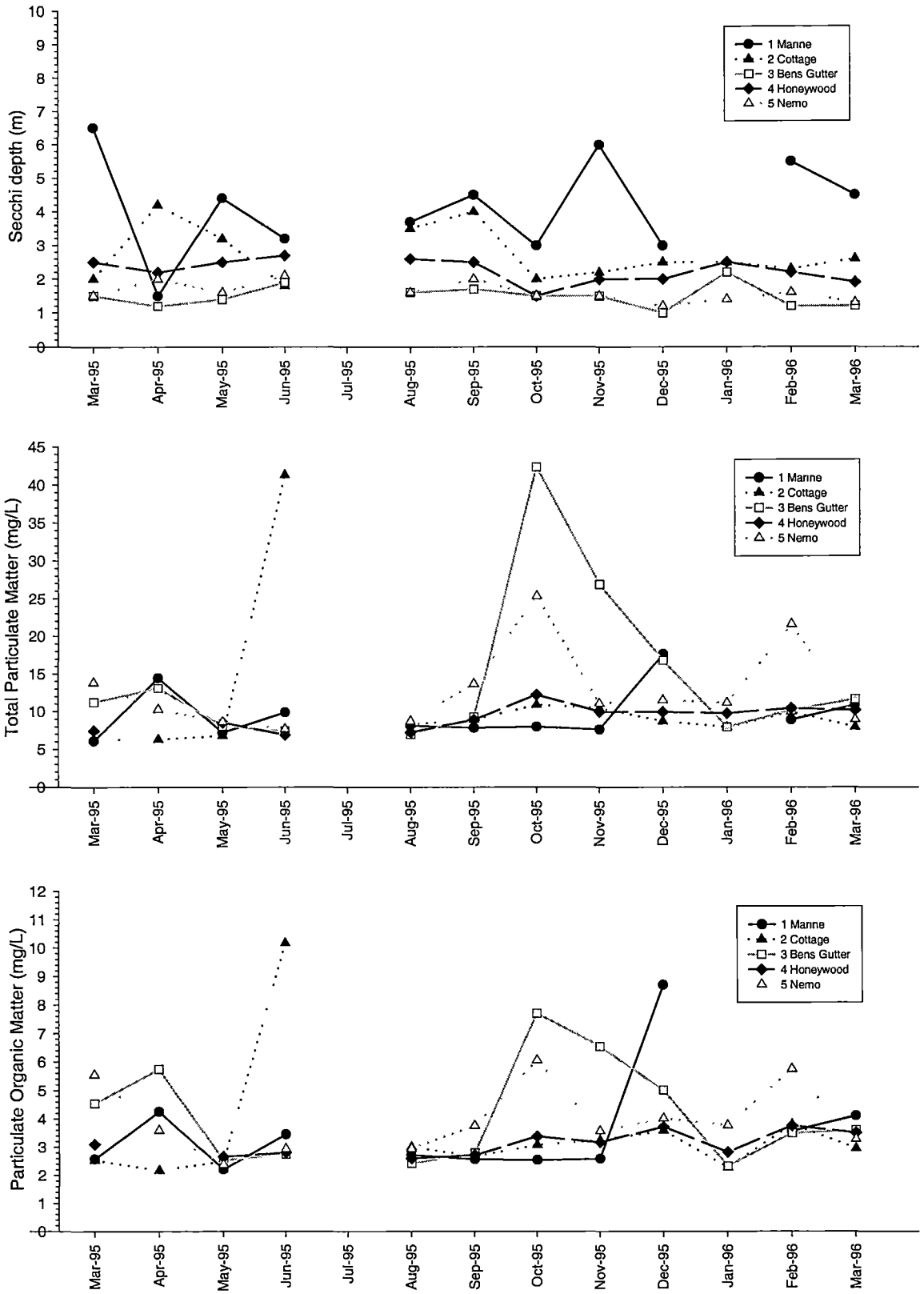

Fig. 2.10 a, b and c. Secchi depth, total particulate matter and particulate organic matter concentrations in Pipeclay Lagoon.

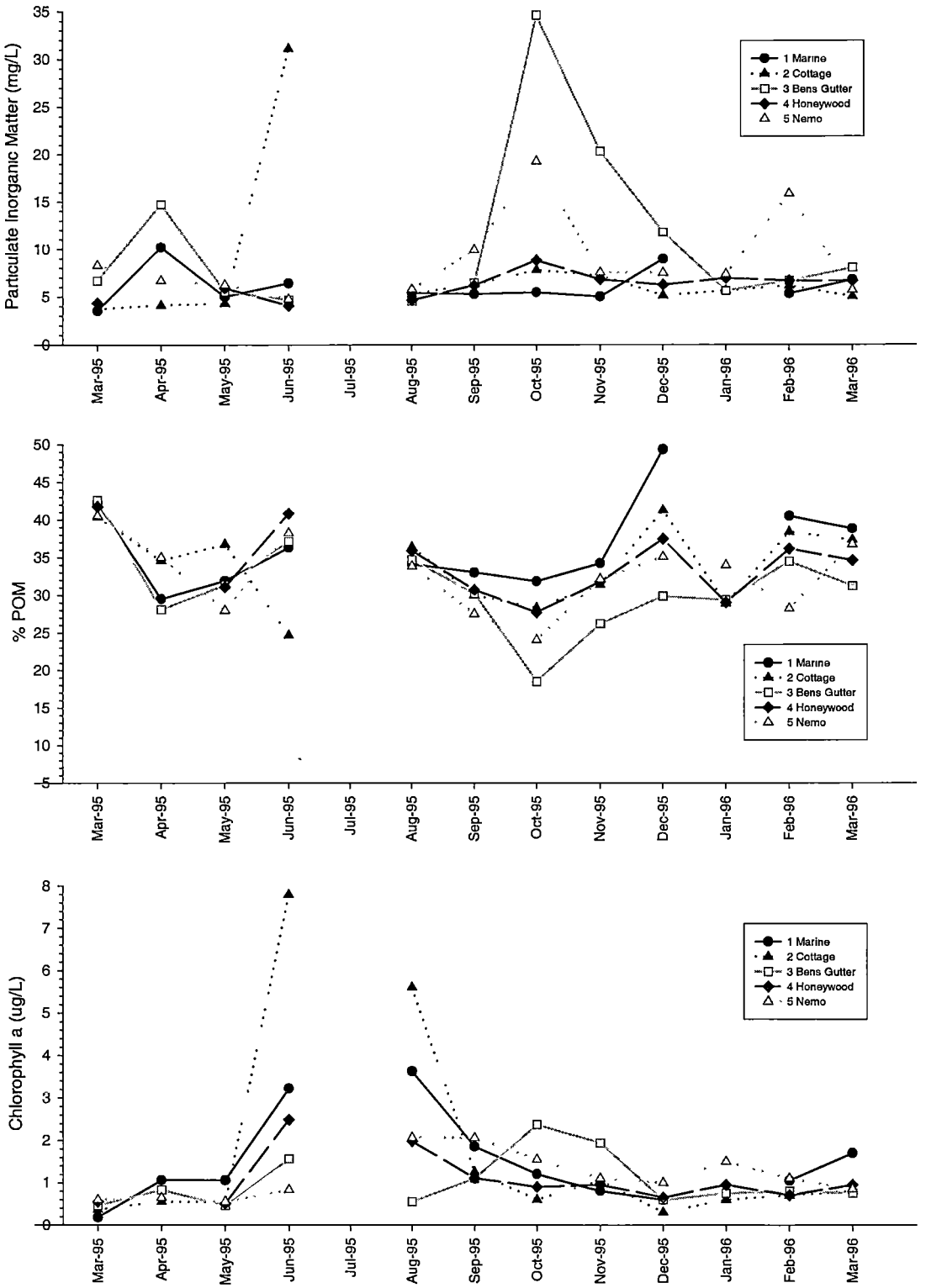

Fig. 2.11 a, b and c. Particulate inorganic matter, percentage particulate organic matter and chlorophyll a concentration in Pipeclay Lagoon.

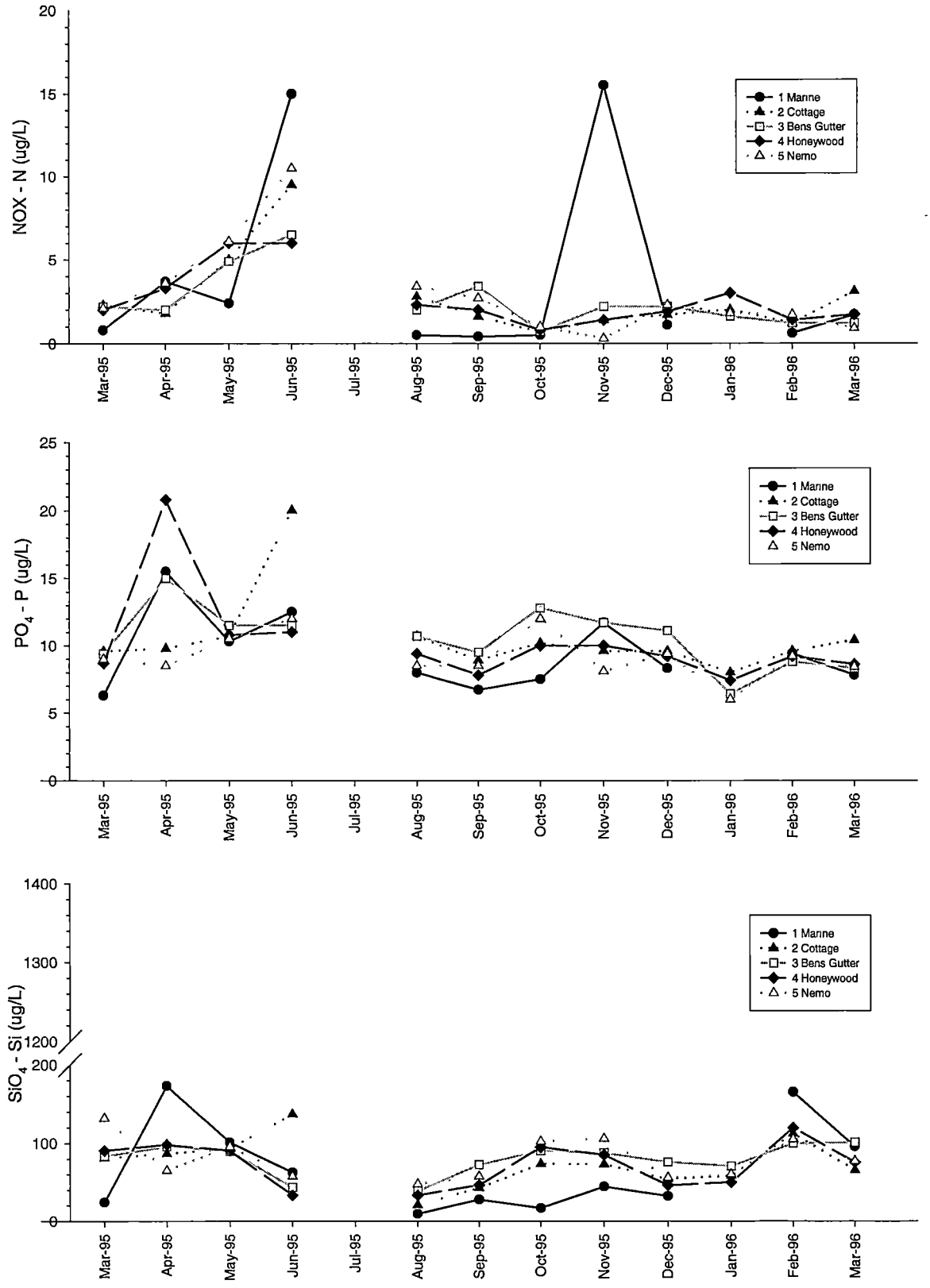

Fig. 2.12 a, b and c. NOX, PO_4-P and SiO_4-Si concentrations in Pipeclay Lagoon.

Little variation or differences in salinity were shown between sites (Fig. 2.9 b). Marginal hypersalinity was apparent within the lagoon on occasions during the warmer periods as compared to the marine station. Minor depression in salinity was shown at sites within the lagoon in January - February 96 following periods of heavy rainfall.

Interestingly, rainfalls during these periods (Fig. 2.9 c) were comparable to those experienced at Pitt Water, though salinities in Pipeclay Lagoon were not so dramatically influenced as occurred in Pitt Water.

The shallow site at Bens Gutter showed little variation in secchi depth and often the bottom was clearly visible (Fig. 2.10 a). Similarly, Nemo and Honeywood sites showed little variation in secchi depths. These sites were also relatively shallow, though on occasions the bottom was not discernible. Only the deeper sites at Cottage and Marine showed changes in secchi depth, or temporal trends. Secchi depths at the Marine site, which was in 6 - 8 m of water, were quite variable and no clear trend apparent. This site was often subjected to swell conditions at time of sampling.

2.3.3.2 Seston quality and *quantity*

TPM was low with occasional elevated levels at some sites within the lagoon (Fig. 2.10 b). Greatest variation occurred at the shallow sites of Bens Gutter and Nemo, most likely due to resuspension of sediments. POM concentrations were within the range of 2 - 6 mg L^{-1} (average 3.68 mg L^{-1}) and similarly showed greater variation at these same sites (Fig. 2.10 c). The highest PIM levels were also measured at these shallow sites (Fig. 2.11 a).

%POM was variable and within the range 30-44% (Fig. 2.11 b). A seasonal trend was shown in %POM with higher levels in winter, decreasing in spring and increasing in summer. Often the Marine site recorded the higher levels.

2.3.3.3 Phytoplankton biomass

Increased chlorophyll a levels occurred in the colder period of June and August with higher levels shown at the Marine site (Fig. 2.11 c). High levels were measured at Cottage during this same time, most likely attributed to uptake of resuspended epiphytes or benthic phytoplankton, from the seagrass (*Heterozostera tasmanica*) beds in this region of the lagoon, though this trend was not consistent. This site was located in the

narrow channel leading into the lagoon, a site which is subjected to strong current flows and frequent boat traffic. Chlorophyll a levels in Pipeclay Lagoon were relatively low and in the range 0.1 - 2.5 $\mu g\ L^{-1}$.

On occasions high chlorophyll a values corresponded to low %POM values as shown in particular at Cottage in June and Bens Gutter in October and November with the greater component of the POM fraction made up of phytoplankton.

2.3.3.4 Nutrients

NOX levels were comparable to those observed at Pitt Water, though slightly higher levels occurred during the cooler months of April to September at sites within the lagoon, with highest levels during June (winter) (Fig. 2.12 a). The Marine site also had elevated NOX concentrations at this time with a high peak in November. Low values were shown during the warmer months. Often the NOX concentration was lower at the marine site than within the lagoon. This could be attributed to other sources, from surrounding regions within the lagoon (e.g. septic seepage).

No clear trend was apparent in phosphate concentrations, though on occasion reduced levels at the marine site coincided with increased chlorophyll a, most likely due to biological uptake (Fig. 2.12 b). Elevated peaks occurred at Honeywood in April and Cottage in June, the latter peak coinciding with a high chlorophyll a reading at this same site. Similarly to NOX, often PO_4-P concentrations were lower at the marine site than within the lagoon. The reduction in levels in January 96 coincided with the time the lagoon was influenced by freshwater inputs following heavy rains and marginal reduction in salinities.

Silicon concentrations at the marine site were comparable to those measured at Pitt Water marine site (Fig. 2.12 c). However, concentrations measured within the lagoon were much lower than those within Pitt Water estuary. A trend of reduced silicon concentrations with increased chlorophyll a is shown, particularly at the Marine site suggesting biological uptake. Higher silicon concentrations occurred within the lagoon, most likely attributed to the shallow nature of the lagoon and resuspension of sediment on occasion.

2.3.4 Little Swanport

2.3.4.1 Temperature, salinity and *secchi depth*

A seasonal trend was shown with temperatures, though extremes of temperature were not so variable as occurred at the other study areas (Fig. 2.13 a). The upper estuary site (Dyke) showed lowest and highest temperature extremes from summer to winter. Temperature differences between the marine and within estuary sites were approximately 4^0 C on most occasions.

Salinity at the Marine site was relatively constant at approximately 35 ppt (Fig. 2.13 b). Salinities within the estuary were reduced in August, November and December following rainfalls. However, salinity within the estuary was only marginally reduced in December despite very heavy rainfalls prior to sampling (Fig. 2.13 c), with reasonable recovery shown in January. The salinity was significantly depressed in February 96 following heavy rains in late January to early February 96. This dramatic decrease in salinity most likely was due to soil saturation and increased catchment run-off following the subsequent rainfalls.

Secchi depths at the Marine site, which was in approximately 10-12 m of water, showed an approximate seasonal trend with shallower depths in the warmer months and deeper depths in the cooler months (Fig. 2.14 a). A similar trend was shown at Plentiful, which was in approximately 6-8 m of water, though secchi depths were shallower and less variable.

Little variation was shown in secchi depths at the upper estuary site (Dyke) which was in approximately 4-5 m of water. The water at this site was observed to be quite turbid with silt/clay suspension. Shack and Jacks Island sites were quite shallow and often the bottom was clearly visible. Secchi depths of approximately 1 m were recorded at all estuary sites in February, coinciding with reduced salinities following flooding of the estuary.

2.3.4.2 Seston quality and *quantity*

TPM was within the range 5 - 10 mg L^{-1} at all sites on most occasions (Fig. 2.14 b). Dyke site showed slightly elevated levels following freshwater inflow during August and November, though no apparent change was shown at the other estuary sites.

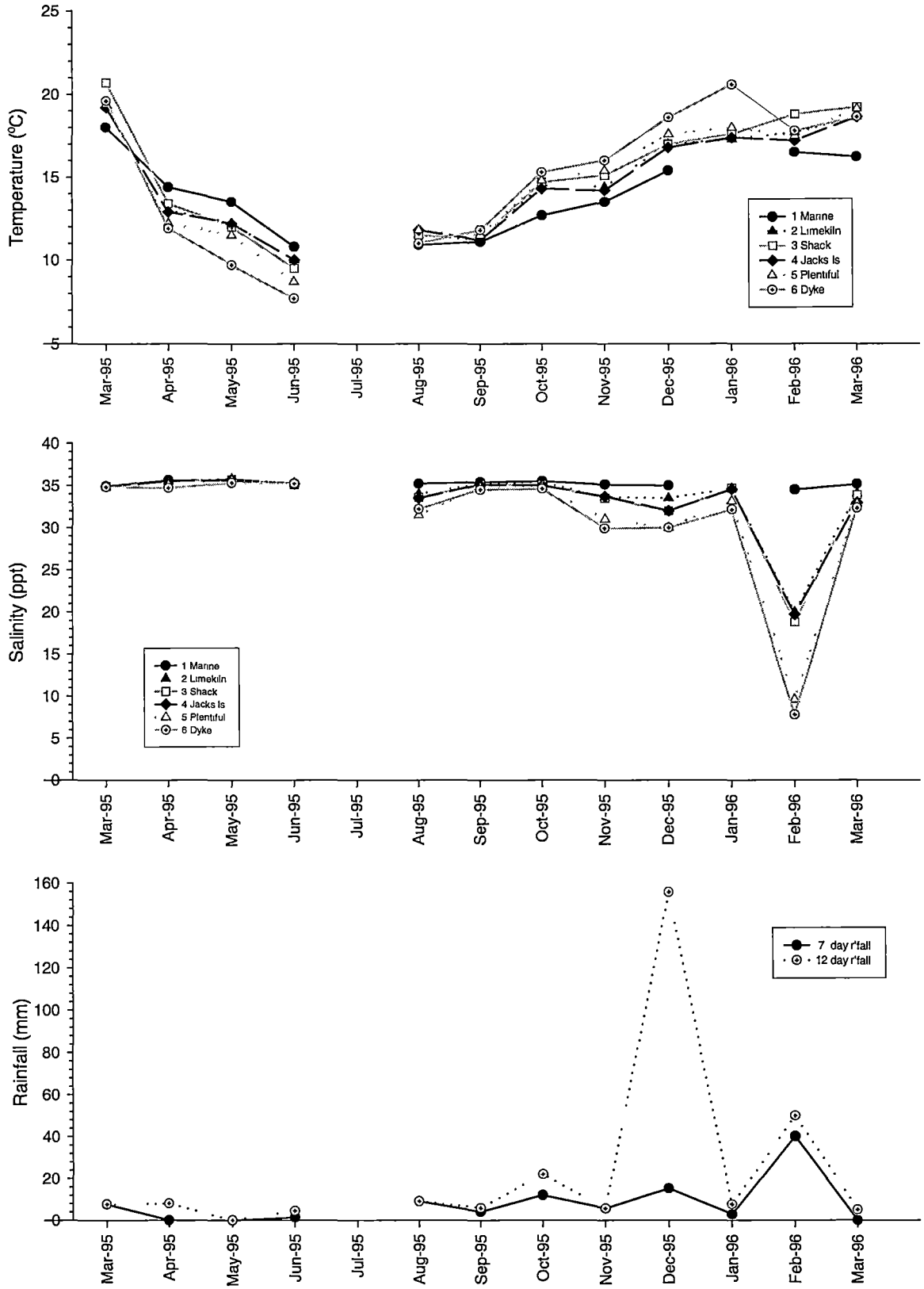

Fig. 2.13 a, b and c. Temperature, salinity, 7 and 12 day cumulative rainfall in Little Swanport.

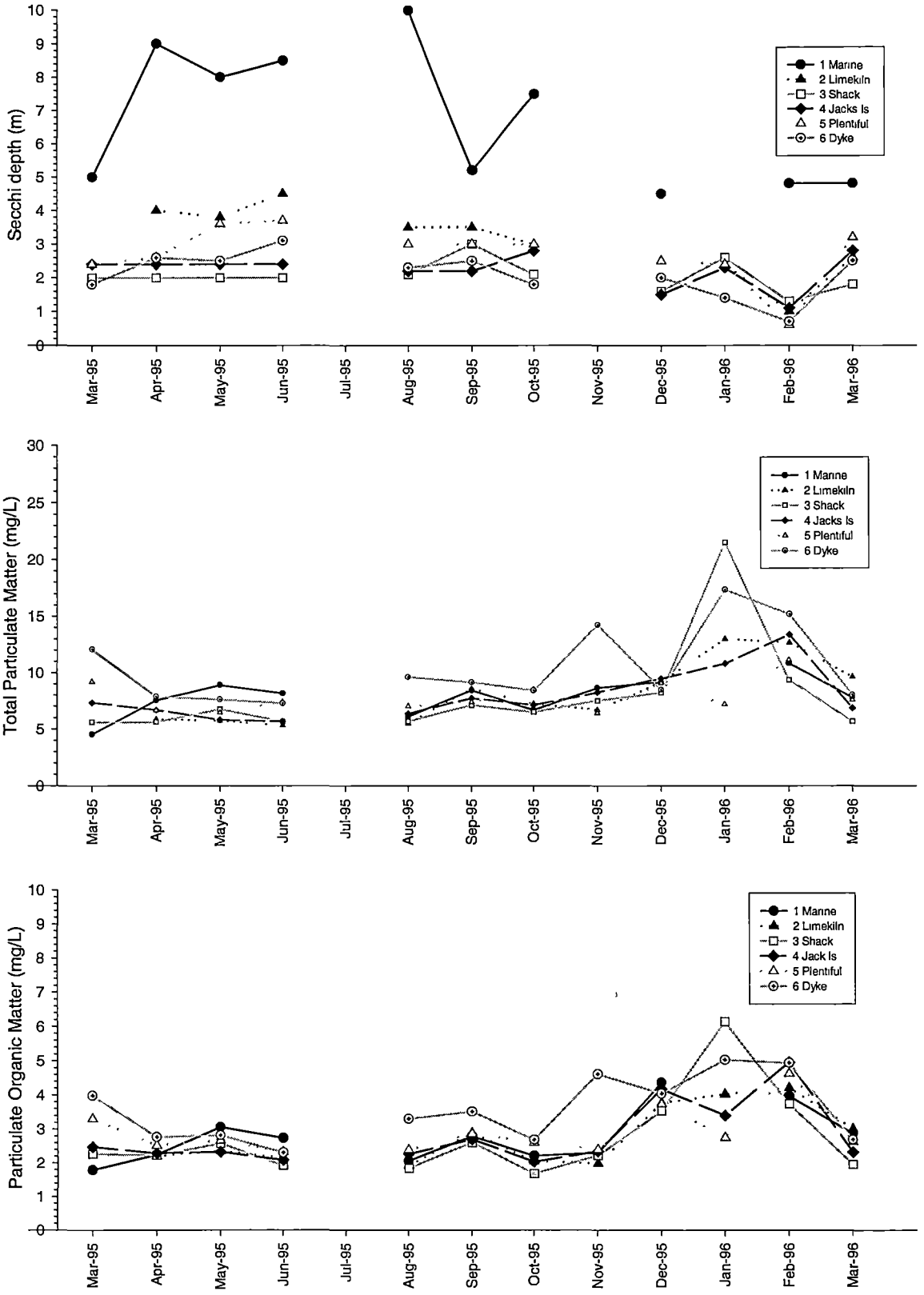

Fig. 2.14 a, b and c. Secchi depth, total particulate matter and particulate organic matter concentrations in Little Swanport.

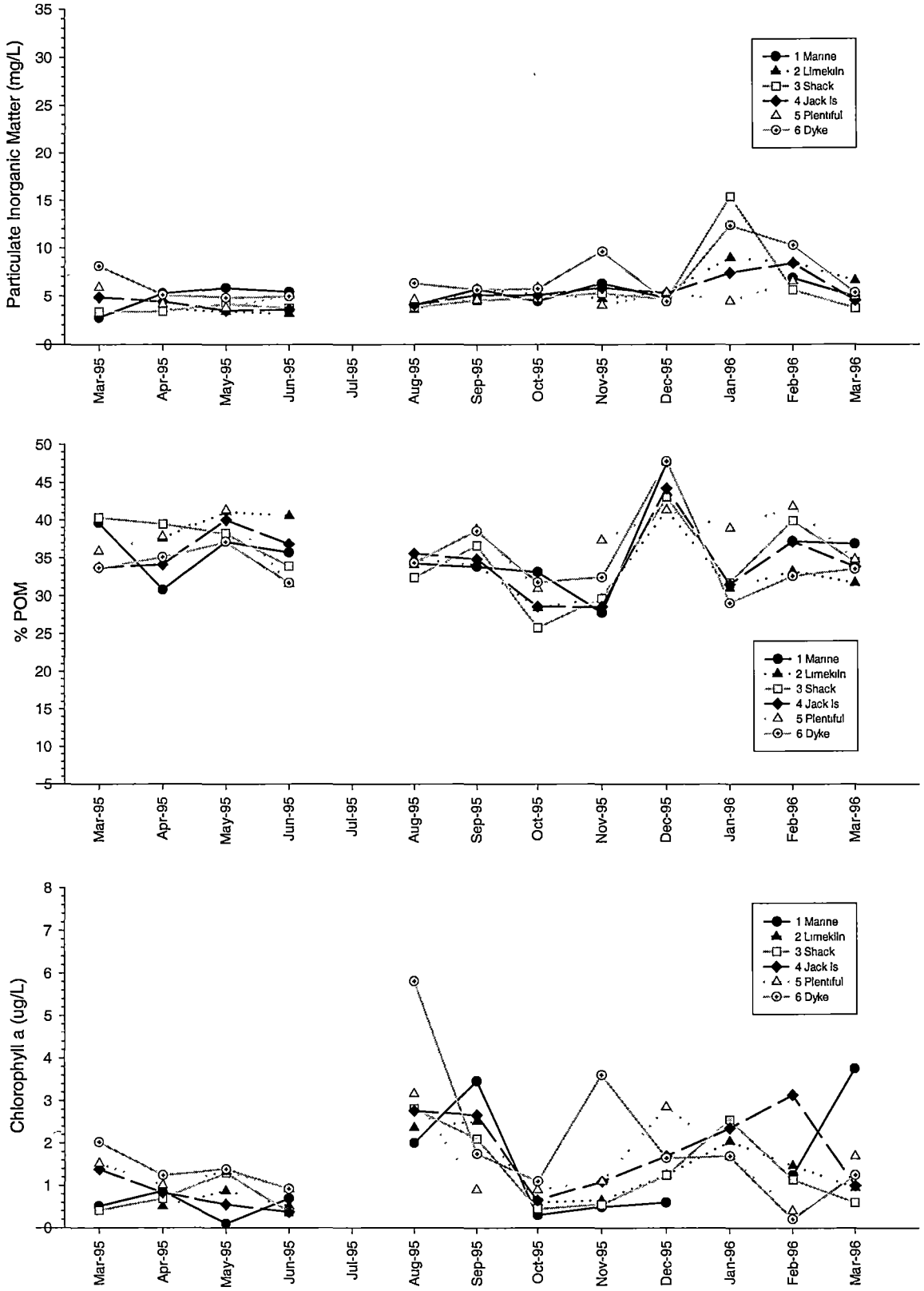

Fig. 2.15 a, b and c. Particulate inorganic matter, percentage particulate organic matter and chlorophyll a concentration in Little Swanport.

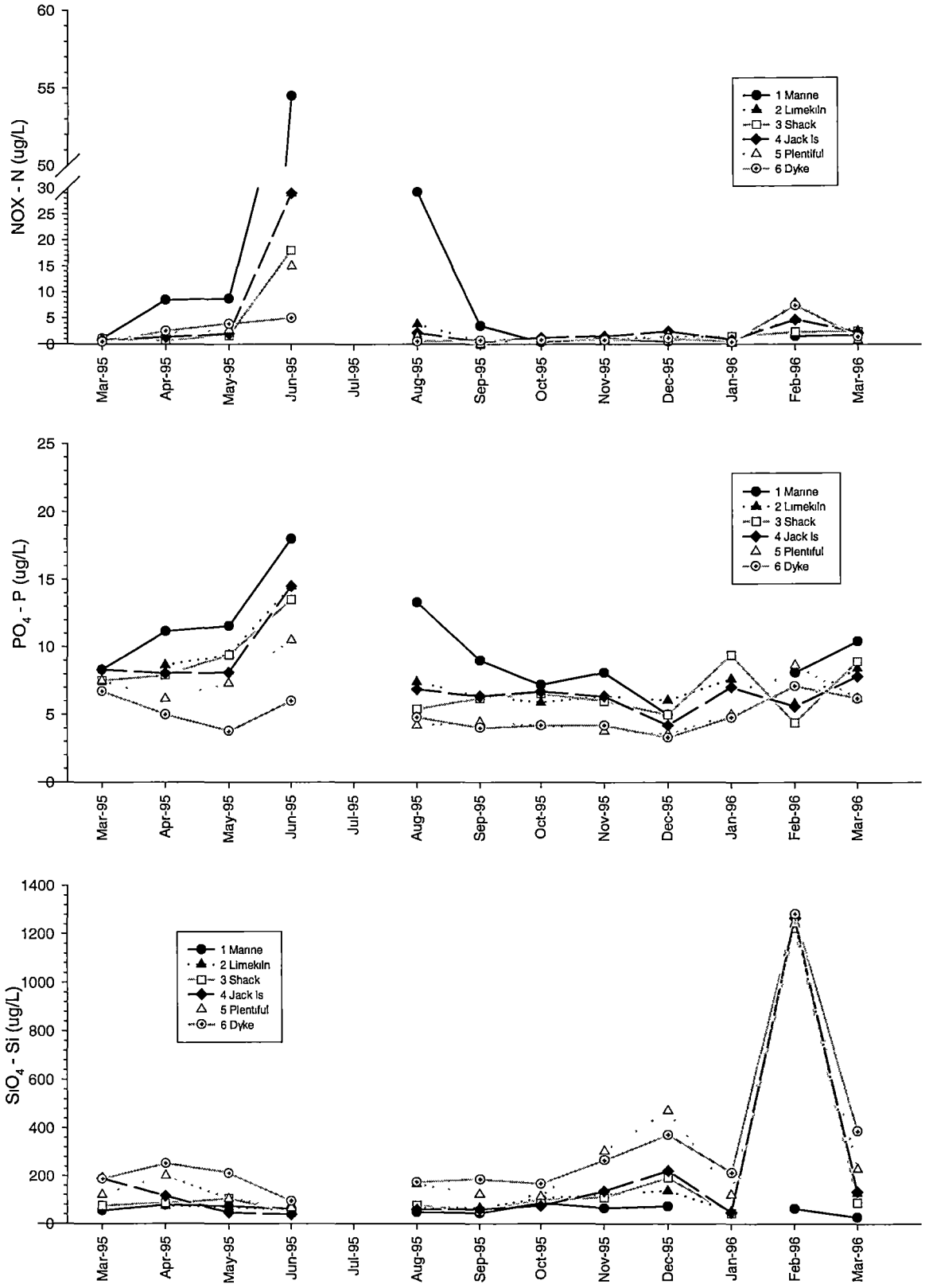

Fig. 2.16 a, b and c. NOX, PO_4-P and SiO_4-Si concentrations in Little Swanport.

The elevated TPM levels at most sites in January coincided with strong winds (25-35 knots) at time of sampling. The marine site could not be reached on this occasion because of rough seas. The elevated TPM levels in February 96 coincided with depressed salinities and flooding of the estuary at that time. Interestingly, TPM levels recorded during December were similar at all sites and low despite heavy rains prior to sampling. Salinities at this time were marginally depressed as compared to February 96.

POM levels were within the range 2 - 5 mg L^{-1} (Fig. 2.14 c). Levels increased in November to March 96 following flooding of the estuary and depressed salinities. PIM levels were low and within the range 3 - 9 mg L^{-1} with the exception of higher levels in January and February 96 (Fig. 2.15 a). %POM levels were within the range of 30-40% (Fig. 2.15 b). The highest levels occurred in December following flooding of the estuary, most likely due to input of organic matter from catchment run-off, though TPM concentrations at this time were not elevated. Levels recorded in February were similar to previous measurements despite flooding of the estuary and significant reductions in salinity.

2.3.4.3 Phytoplankton biomass

The estuary sites, in particular the upper estuary sites, had higher chlorophyll a levels than the marine site, with the exception of September and March 96 (Fig. 2.15 c). A seasonal trend was apparent with higher chlorophyll a concentrations in late winter to early spring, decreasing in spring and increasing in summer. Elevated levels recorded at Dyke in August and November coincided with rainfall run-off prior to sampling and reduced salinities. However, this was not the case in February despite heavy rains and depressed salinities.

2.3.4.4 Nutrients

Elevated NOX levels were recorded in June at all sites, with the exception of Dyke (Fig. 2.16 a). The Marine site had the highest peaks in the winter months of June (54 ug L^{-1}) and August (30 ug L^{-1}), though levels recorded at the estuary sites during August were low. Low concentrations were recorded at all sites from October to January, but never fell below non-detectable levels. A marginal increase in February occurred during the time of freshwater inflow and flooding of the estuary, though this was not observed on the previous occasions during reduced salinities.

A trend of decreasing PO_4-P concentrations progressing up the estuary was noted, the Marine site recorded the highest levels and the upper estuary site (Dyke) the lowest (Fig. 2.16 b). Higher values were measured in the cooler months with no notable change following flooding of the estuary, particularly in February. A trend was apparent between chlorophyll a and PO_4-P, with reduced PO_4-P levels coinciding with increased chlorophyll a possibly due to biological uptake.

Silicon levels at the Marine site were consistently lower than the estuary sites and showed little variability (Fig. 2.16 c). The trend of increasing SiO_4–Si concentrations progressing up the estuary was shown, with the upper estuary sites (Dyke and Plentiful) recording the higher levels. Elevated SiO_4–Si concentrations were measured at Dyke and Plentiful in November and December, coinciding with reduced salinities and fresh water inflows. The highest levels at all estuary sites occurred in February 96, when the estuary flooded.

2.4 Discussion

Three of the major oyster growing areas within Tasmania – Pitt Water, Pipeclay Lagoon and Little Swanport – were sampled over a 13 month period to assess water column parameters with the view to determining variations within each area, and hence factors which may be responsible for influencing food quality and quantity and hence shellfish productivity.

The pilot study conducted at Pitt Water showed variability at spatial and temporal scales in the parameters measured over days and between weeks within a month at two sites approximately 3 km apart. Rough sea conditions were encountered on the sampling days in the later 3 weeks of the study, with 15 to 30 knots winds predominantly from the north–west or south–west direction. Pitt Water estuary is relatively shallow and during such conditions rapidly becomes quite turbid, due to resuspension of sediments (pers. obs.). Additionally, hydrodynamic changes occur as the result of wind driven currents. The upper estuary site at Shark Point showed the marginally higher recordings for temperature, salinity, PO_4–P, and SiO_4–Si on most occasions. Chlorophyll a levels were marginally higher at the North Causeway site from day 6 to the end of the four week period. The variation in prevailing weather conditions during the sampling period, most likely contributed to the spatial and temporal variability in the parameters measured.

The variability in each of the parameters measured at the two sites in Pitt Water over 24 hours was similar to the range measured over the month (Crawford et al., 1996). This study showed that frequent sampling over tidal cycles, or daily, would be needed to study factors influencing water column processes in more detail. Nevertheless, sampling at time intervals of a month for long periods (12 months → years) does provide information on spatial and temporal trends due to seasonal, or other influences (such as flood events, pollution inputs, alterations to hydrology). These sampling schedules frequently encompass most conditions encountered within an area (broadly classed as 'dry' or 'wet' conditions). However, targeted sampling can be conducted on occasions to cover special events of interest, such as during periods of heavy rainfalls, extended dry conditions or windy periods.

Changes were observed in most of the parameters measured, with many following an apparent seasonal pattern. However, the use of calendar months for categorising seasons is questionable since natural cycles may deviate from this rigid temporal pattern. Harris et al. (1987) showed that the timing of the spring bloom at an inshore station near Maria Island (east coast of Tasmania) varied from year to year by as much as 4 months. Means by which this may be elucidated are complex, and were not investigated in this study. The seasonal pattern in temperate regions of minimum temperatures in the winter months and maximum temperatures in the summer months was shown. Pipeclay Lagoon and Pitt Water showed colder winter temperatures than Little Swanport. Differences in temperature between the marine and within estuary/embayment sites were shown at each area. The three areas studied were relatively shallow well mixed systems, and many of the sites sampled were in shallow water. The differences in temperatures between the marine stations was due to the influence of ambient temperature conditions (either cooling or warming the water mass) at the within estuary/embayment sites.

Compared to the marine stations, higher salinities due to evaporation occurred at Pitt Water and Pipeclay Lagoon, though not as marked as has previously been recorded at these sites (Crawford and Mitchell, 1999). Pitt Water and Little Swanport were more influenced by freshwater inflows than Pipeclay Lagoon. Although Pipeclay Lagoon experienced similar rainfall intensities, salinity altered little with a marginal decrease in salinities recorded within the lagoon and rapid recovery to 'marine' levels. This could be attributed to either the catchment absorbing much of the rainfall (particularly the marsh regions located north and south of the lagoon), or a more rapid flushing of the

lagoon with marine water from Frederick Henry Bay. Most heavy rainfalls occurred in the later part of the year during December - January, generally following historical patterns. The degree by which each area was influenced by freshwater inputs (or flooding) and the rate of recovery was variable. Pitt Water showed slow recovery, suggestive of reduced flushing or exchange capacity.

Cumulative rainfall within the previous 7 days correlated well with reductions in salinity, though on occasions rainfall within the previous 12 days provided a better indication. A correlation in 7 day cumulative rainfall and depression in salinity has been found in previous studies of shellfish growing areas around Tasmania (Tasmanian Shellfish Quality Assurance Program) and has been used to monitor conditions likely to cause elevated faecal coliform counts and hence closure for the harvesting of shellfish for sale (e.g. Mitchell, 1988; Mitchell and Brown, 1992). Catchment runoff, and hence flooding, is linked largely to soil saturation within the catchment. During the period of this study, moderate rainfalls were recorded in the catchments of each area with little evidence of reductions in salinities, and hence freshwater inflows. However, on occasions subsequent rainfalls which increased soil saturation resulted in catchment runoff leading to reductions in salinity, particularly at Pitt Water and Little Swanport, but not so apparent at Pipeclay Lagoon.

Seston (or total particulate matter) concentrations were similar between Pitt Water and Pipeclay Lagoon, with Little Swanport showing lower values. On occasions in each area, higher recordings were apparent at shallow sites frequently attributed to resuspension of sediment, either due to active tidal flows or wind driven turbulence. At times, sample bottles were blown, or transported by tidal flows, into shallower water which resulted in sediment uptake. Whilst these results were removed from the data analysis, they did provide useful information on the sediment quality, in particular benthic phytoplankton abundance (discussed later). The higher levels were generally recorded in the upper estuary sites of Little Swanport (Dyke), Pitt Water (Top End) and Pipeclay Lagoon (Nemo).

TPM concentrations at Little Swanport and Pipeclay Lagoon, average 8.23 ± 2.66 and 12.06 ± 8.70 mg L^{-1} respectively, were higher than those measured by Brown and McCausland (1999). However, the values reported by these authors were for the <20 μm fraction of the seawater filtered, whereas values reported here were for the <500 μm fraction. Differences could also be due to differing methods, with respect to volumes

filtered and temperatures used to dry and ash filters. Hawkins et al. (1998), for example, reported lower TPM levels measured in seston samples which had been oven dried at 110^0 C as compared to those dried at 60^0 C.

Whilst seston concentrations were comparable within each area, differences were evident in the quality, as determined by percentage particulate organic matter. Higher %POM levels were recorded at each marine site. Little Swanport had the higher %POM fractions at all sites within the estuary on most occasions (mean range 34.2–37.1). Pipeclay Lagoon had similar levels, with %POM generally above 30%. Pitt Water tended to show marginally lower %POM levels, particularly within the upper estuary region (mean range 22.1–29.9), with the upper most site (Top End) having the lowest levels. This most likely was due to the suspension of silt/clay frequently observed in the water column in Upper Pitt Water. This inorganic material at times formed a greater component of the seston than phytoplankton or organic matter. On occasions, higher chlorophyll a concentrations corresponded to low %POM values, and suggest that a large component of the organic matter may have been due to phytoplankton. Elevated %POM levels were also observed at times due to freshwater inflows at Pitt Water and Little Swanport, most likely due to inputs of organic matter from catchment runoff.

Secchi depths only proved useful in the deeper sites (i.e. greater than approximately 2 m depth) within the three areas studied. Several sites were quite shallow, and whilst the bottom was not clearly discernible, the secchi disk was visible. This was most apparent at the estuary sites of Pitt Water (Woody Island and Barilla), Little Swanport (Limekiln, Shack and Jacks Island) and Pipeclay Lagoon (Bens Gutter, Honeywood and Nemo). Greater ranges in secchi depths were measured at the marine sites at each area, with less variation shown at the within estuary/embayment sites. No discernible trend was evident with secchi depths, with the exception of Lewisham at Pitt Water. This site showed the trend of greater secchi depths in the cooler months and shallower depths in the warmer months. The upper estuary sites of Little Swanport (Plentiful and Dyke) and Pitt Water (North Causeway, Barilla, Shark Point and Top End) showed little variation in secchi depths, generally attributed to the more turbid water observed on each sampling occasion. Secchi depths have been found to be not related to primary production if turbidity is influenced by other factors (Vollenweider, 1969). They do, however, provide a simple means of gaining additional information to assist in the interpretation of water column parameters measured.

Chlorophyll a concentrations in each area were low, ranging from 0.2 to 4.0 $\mu g\ L^{-1}$. Pipeclay Lagoon showed the lower chlorophyll a concentrations with those recorded at Pitt Water marginally higher. Chlorophyll a concentrations at Little Swanport were the highest. The mid to upper estuary sites at Pitt Water showed higher concentrations as compared to the Marine and Lewisham stations. On occasions high chlorophyll a concentrations were measured in samples where sediment uptake had occurred. Whilst these were omitted from the data analysis as outliers, they did suggest the presence of high benthic phytoplankton populations. This observation was made at Woody Island (Pitt Water), and at Cottage and Bens Gutter in Pipeclay Lagoon.

Interestingly, chlorophyll a levels measured were high in the winter to early spring months within each area, which is in opposition to the normal pattern in temperate systems of higher concentrations in the spring/summer period and low concentrations in the cooler months. Higher levels were recorded at the estuary sites of Pitt Water following the significant rainfalls in the latter part of 1995 and early 1996 and suggest a beneficial effect of the freshwater inflows. These levels corresponded to reduced salinities at these sites during this same time period. This trend was not so apparent at Little Swanport which experienced similar rainfall intensities, and may have been attributed to flushing of the estuary, a phenomenon which has been observed in the past with bacterial loads (Mitchell, 1988).

Brown and McCausland (1999) found average chlorophyll a concentrations in Little Swanport to be approximately double those of Pipeclay Lagoon, with approximately 75% of the total chlorophyll a contained in the <20 μm fraction. They also found differences in the predominant phytoplankton species composition, as shown by HPLC pigment analyses. Van den Enden (1994) found chlorophyll a concentrations in Little Swanport to vary according to phytoplankton abundance, with >90% of the 116 phytoplankton species identified comprising diatoms, many of which were of benthic origin. This was also shown by the presence of fucoxanthin pigment (characteristic of diatoms) which was always present in the water samples. High levels of carotenoid pigments were also recorded on occasion and indicated increased abundance of non–living phytoplankton or detritus. Van den Enden (1994) indicated, from pigment analysis of oyster stomach contents, that detritus from seagrass (*Zostera sp*) provided an additional food source to oysters. A similar study conducted by Hallegraeff et al. (1986) showed the predominant phytoplankton species composition in Pitt Water to be diatoms,

with numerous benthic species resuspended from the sediments. These authors noted the phytoplankton population within Pitt Water to be quite distinct from Storm Bay phytoplankton.

Microphytobenthic biomass has been shown to be an important source of primary production in shallow tidal embayments, and a significant component of chlorophyll a in the water column during resuspension of sediments, either due to wind action or tidal flows (Colijin and Dijkema, 1981; Lukatelich and McComb, 1986; Barranguet, 1997; Guarini et al., 1998). Benthic diatoms were shown to form a significant component of oyster diets in Little Swanport (van den Enden, 1994) and Pitt Water (Hallegraeff et al., 1986). Similarly, it has been suggested that microphytobenthos forms a significant food source of oysters and mussels in Marennes–Oléron Bay (France) (Pastoureaud et al., 1996; Smaal and Zurburg, 1997).

NOX concentrations were low (less than 4 – 10 $\mu g\ L^{-1}$) at each site, with the exception of the marine sites at Little Swanport and Pipeclay Lagoon in the winter months and Pipeclay Lagoon in November 1995. Elevated levels were recorded at the estuary sites of Pitt Water in February and March 1996, coinciding with reduced salinities as a consequence of freshwater inflows. A high peak was recorded at Lewisham, most likely due to localised freshwater inflow from the surrounding catchment and/or seepage from septic systems (a common problem in this region because of the sandy soil and close proximity of houses along the foreshore). Ortho–phosphate concentrations were low, and within the range 6 – 13 $\mu g\ L^{-1}$ at most sites. However, elevated levels were recorded at Little Swanport and Pipeclay during the winter months when NOX concentrations were also high. Little Swanport showed the trend of decreasing PO_4-P levels from the marine site to the upper estuary site, though no similar distinct trend was apparent at Pitt Water or Pipeclay. The higher levels recorded at Woody Island from January – March 1996, coinciding with reduced salinities, may have been attributed to outflow of water from Orielton Lagoon. High levels of total phosphorus and ortho–phosphate have been recorded in the sediments and water of this lagoon (Kinhill, 1993). A trend of reduced NOX and PO_4–P at times of elevated chlorophyll a was observed within each of the study areas and suggests biological uptake of these nutrients.

Silicon levels were reasonably similar at each of the marine sites, with a reduction shown during periods of elevated chlorophyll a levels, suggesting biological uptake presumably by diatoms. Low concentrations were shown in Pipeclay Lagoon, with

higher levels measured at Pitt Water and Little Swanport. Within these two areas, a trend of decreasing concentrations was shown seaward along the estuary. Frequently, elevated levels were recorded at the more shallow sites due to resuspension of sediments. High levels were recorded during reduced salinities and freshwater inputs during the later sampling months at Pitt Water and Little Swanport.

Sampling during this study was fortunate to cover a relatively broad range of environmental conditions, in particular heavy rainfall periods. The effects of these freshwater inflows was most pronounced at Pitt Water and Little Swanport, with little influence apparent at Pipeclay Lagoon. Elevated nutrient and chlorophyll a concentrations occurred as a consequence of these events and were more pronounced at Pitt Water, which showed reduced flushing rates, as evidenced by the salinity measurements. In recent times much interest has been shown in Pitt Water as a consequence of the Craigbourne Dam, and subsequent weir constructed below the Richmond weir in 1992. A similar study conducted by Crawford and Mitchell (1999) showed elevated NOX levels pre mid-1992, and reduced concentrations subsequently. Tyler et al. (1986) reported reduced salinities at Pitt Water following rainfall events prior to the dam construction in 1986, however such events were not so frequent after the construction of the dam, and suggested a buffering effect. Similarly, Brown and Mitchell (1992) reported reduced flood events post dam construction, and supported the observation of the buffering effect of the dam.

In Pitt Water following the flooding experienced during this study, NOX levels were elevated with increased chlorophyll a concentrations as well as greater fractions of %POM were measured. The observation of elevated NOX levels concurrently with increased chlorophyll a within the upper region of Pitt Water suggests that other factors limited phytoplankton growth during this period. It would be expected that increased NOX levels would promote algal growth, however, this was not shown during this time and suggests that turbidity was most likely responsible for limiting algal growth. Little Swanport showed a similar trend, however greater flushing of the estuary was evident, with reduced levels of NOX and chlorophyll a as a consequence, though recovery was reasonably rapid.

Turbidity has been strongly linked as a principal factor limiting phytoplankton growth, rather than nutrient limitation (e.g. Joint and Pomroy, 1981; Cloern, 1987). Elevated levels of total particulate matter as a result of riverine inputs or resuspension of

sediments, has generally been found to be the cause of high turbidity. Resuspension of sediments occur as a consequence of wind action, though tidal currents also cause this to happen, with the degree of continued state of resuspension a factor of particle size (Day, 1981a). This has been shown to cause variable measures of phytoplankton abundance spatially and temporally (Therriault and Platt, 1981; Litaker et al., 1993; Geyer, 1997). However, mixing of the water column can still promote phytoplankton growth (Fichez et al., 1992). Sediments in the upper estuary region of Pitt Water are predominantly finer than the lower estuary region (Mitchell et al., 1998), with a similar observation noted at Little Swanport. Pipeclay Lagoon sediments are predominantly a medium to fine sand (Mitchell and Macleod, 1998). Water clarity in the upper estuary regions of Pitt Water and Little Swanport frequently has been observed to be reduced due to the suspension of fine particulate matter/sediment (pers. obs.).

Pitt Water estuary is frequently exposed to winds of sufficient strength to cause resuspension of sediments. An assessment of wind data records for Hobart Airport for the period 1958–98 showed the prevailing wind direction to be from the west to north sector (54%) (generally down the estuary), and the average percentage frequency of wind speed over a 24 hr period was calculated as 59% > 11 km/hr (Mitchell et al., 1998). Pipeclay Lagoon has also been observed to have reduced water clarity, predominantly due to wind driven resuspension, though generally water clarity improves more rapidly than in Pitt Water as a consequence of the more rapid settlement of suspended sediments.

Studies have been conducted to determine if light or nutrients limit phytoplankton production within different regions along estuaries (e.g. O'Donohue and Dennison, 1997) or at different times of the year (e.g. Pennock and Sharp, 1994). O'Donohue and Dennison (1997) found that phytoplankton productivity was stimulated by phosphorus enrichment at upriver sites, and by nitrogen additions at the lower river/bay sites. They also found that light stimulation was more pronounced at upriver sites, which were more turbid, than the lower river/bay region.

During the colder months, in particular June, elevated peaks of NOX were recorded at the marine station of Little Swanport and Pipeclay Lagoon, with Little Swanport recording high levels in June and August (54 and 30 $\mu g\ L^{-1}$ respectively). A peak was observed at Pipeclay during these months and in November, but no secondary peak was observed in November at Little Swanport. Similarly, high nitrate concentrations have

been recorded during the months of March to September near Maria Island on the east coast (Harris et al., 1987) and Storm Bay (Clementson et al., 1989; Harris et al., 1991). These events are linked to the movement of complex water masses around the south–east and east coast of Tasmania, which significantly influence the seasonal and interannual variability in nutrients and phytoplankton biomass (Harris et al., 1987; Clementson et al., 1989; Harris et al., 1991). The movement of these water masses is largely due to El Niño–Southern Oscillation (ENSO) events (Hsieh and Hamon, 1991).

Variations in temperature, salinity and nutrient concentrations have been used to characterise the different water masses of the subtropical (ST), subantarctic (SAW) and subtropical convergence (STC) which form the predominant complex mixture of waters in south–eastern Tasmania (Harris et al., 1991). Clementson et al. (1989) showed the water in Storm Bay to be influenced by water of subtropical origin from the east and water of subantarctic origin from the west, with the predominant water mass varying at different times of the year. These authors showed that chemical and biological parameters measured differed between and within years, and the timing and duration of phytoplankton blooms were strongly linked to the westerly wind stress. High nitrate concentrations were associated with the strong influence of SAW during the cooler winter period, with subsequent high phytoplankton biomass. ST water was more prominent during the summer period, and typically had lower nutrient concentrations, particularly DIP. Similarly, a seasonal cycle of increased nitrate measured at the inshore Maria Island station coincided with autumn and winter cooling of the water mass and ended with the spring warming (Harris et al., 1987). However, these authors showed that occasionally high nitrate concentrations occurred during November and December attributed to transport of offshore water over the shelf towards inshore. However, this was not observed at the marine station of Little Swanport during this study, or during the study conducted by Crawford and Mitchell (1999).

Coughanowr (1995) reported a similar pattern of elevated NOX (30 – 50 $\mu g\ L^{-1}$) during late June to early September 1993 at the marine site of the Derwent Estuary. This suggests the water in this region is influenced by the water mass within Storm Bay. It is therefore likely that the marine station at Pipeclay Lagoon is also influenced by the water mass within Storm Bay, though NOX concentrations measured were lower than those recorded by Coughanowr (1995) or Clementson et al. (1989). The elevated NOX peaks observed at the marine site of Pipeclay Lagoon during November of this study and similarly recorded by Crawford and Mitchell (1999), may have been attributed to

injections of nitrate into the stratified surface waters of Storm Bay as a consequence of westerly winds (Harris et al., 1991). A similar phenomenon has been observed in Saldanha Bay (South Africa), where phytoplankton biomass and primary productivity have been strongly linked to the Benguela upwelling system (Monteiro and Brundrit, 1990; Pitcher and Calder, 1998).

This trend was not so apparent at the marine site of Pitt Water, located in the upper reaches of Frederick Henry Bay, and suggests either a dilution effect, diminished flow of this nutrient rich water mass to this region, or biological uptake resulting in low nutrient concentrations measured. Interestingly, elevated chlorophyll a levels were recorded at this site during this time with concurrent reduction in silicon concentrations, suggesting biological uptake most likely by diatoms. A similar pattern of elevated chlorophyll a and NOX was shown at this site by Crawford and Mitchell (1999), though NOX concentrations were less than those recorded at Pipeclay Lagoon marine site.

The hydrographic observations show how these water masses influence nutrient concentrations, and hence the degree by which the timing, magnitude and duration of phytoplankton biomass varies. Given the marine nature of Pipeclay Lagoon, these events would significantly influence food abundance within this embayment and hence oyster growth and condition. A longer term study by Brown and McCausland (1999) also reported seasonal variations in nitrate concentrations within Pipeclay Lagoon, with maximum levels observed from July to September. These authors reported maximal chlorophyll a levels during this same period, with high levels of chlorophylls c1 + c2 (reported to be found in diatoms) measured in Pipeclay Lagoon. This coincides with the observation of the dominance of mostly diatoms during high productivity time periods within Storm Bay (Harris et al., 1991).

These observations are worthy of note with respect to Pipeclay Lagoon, and possibly Little Swanport, in regard to predicting favourable times for oyster growth and condition. Research has shown that these events can, to a large extent, be forecasted. The periodicity of the westerly wind stress observed by Clementson et al. (1989), which influences nutrient cycling and phytoplankton biomass, appears to be one of the generating mechanisms of the ENSO phenomena (Harris et al., 1991). This phenomena has been linked to the Southern Oscillation Index (SOI), measured from barometric changes, which enable prediction of either warm or cold ENSO events (Hsieh and Hamon, 1991).

Results of the water column parameters measured during this study indicate that Little Swanport has the greater seston quality, as compared to Pitt Water and Pipeclay Lagoon. Whilst chlorophyll a levels appeared marginally higher in Pitt Water, seston quality (as measured by %POM) was much lower than that measured in Pipeclay Lagoon or Little Swanport. The upper estuary region of Little Swanport is characterised by extensive mud flats vegetated predominantly by reeds and ricegrass (*Spartinia anglica*), with extensive beds of eelgrass (*Zostera muelleri*) in the mid to lower region of the estuary. Detritus originating from these sources could well account for the higher %POM levels recorded in this estuary, an observation supported by the high carotenoid pigment levels recorded by van den Enden (1994). The lower levels of PO_4–P recorded at the uppermost site in Little Swanport could be attributed to absorption and binding of phosphorus within the marsh/mud flat regions.

It appears that Upper Pitt Water supports a more autochthonous phytoplankton population. This observation is supported by the study conducted by Hallegraeff et al. (1986). The region of the upper estuary showed depressed salinities from November through to March, with slow recovery suggestive of reduced flushing or exchange with marine water. During this time, chlorophyll a levels remained high, and were greater than those recorded at the marine site. Previous sampling of the estuary has generally shown higher chlorophyll a levels in the upper estuary, as compared to the lower estuary and marine site (Crawford and Mitchell, 1999).

Light limitation as a result of increased turbidity in the upper reaches of Pitt Water and Little Swanport is most likely an important factor restricting phytoplankton abundance. The question of whether nutrient limitation occurs is not so clearly answerable. Frequently this has been determined based on the Redfield ratio (C:N:P of 106:15:1) using TN:TP (or often DIN:DIP) (e.g. Coughanowr, 1995; O'Donohue and Dennison, 1997), however some debate exists over the validity and applicability of this (Oliver, 1998). This seems logical, as in most studies nutrients measured are those present at the point in time the sample is collected, and do not provide an adequate means of assessing available nutrients. Low nutrient levels measured do not necessarily indicate limitations on available supplies. It is often cited that nitrogen frequently is the limiting nutrient in marine systems, however there is increasing evidence to show that phosphorus limitation occurs (Koop et al., 1998). These authors highlighted that whilst nutrient concentrations may be low, cycling rates are high as evidenced by continued phytoplankton production at times when nutrient levels were low to undetectable. They

also supported the fact that little information is available on, or support given to, the importance of sediment processes, particularly microbial transformations which re–cycle and re–supply nutrients to the overlying waters. Sediment nutrient dynamics play a key role in water column nutrient processes, and have been shown to be a significant contributor of available nutrients to the overlying waters and hence to phytoplankton (e.g. Pomroy et al., 1983; Nowicki and Nixon, 1985; Simon, 1988; Forès et al., 1994; Feuillet–Girard et al., 1997).

A point of interest is the difference in chlorophyll a concentrations calculated depending on the equation used. A comparison was made of the equation used by van den Enden (1994) and Crawford et al., (1996) with that used in this study, using the data collected. Chlorophyll a concentrations (on average) were approximately 30% lower than those calculated with the equation used by van den Enden (1994) and Crawford et al., (1996). The differentiation of chlorophyll a and its degraded pigments (phaeopigments) using the method of Tett (1987), was intended to enable estimation of the quality of the phytoplankton biomass with respect to the proportion of active (living) phytoplankton as compared to the non–living fraction (i.e. phaeopigments measured by acidification of the extracts). Frequently, values calculated for phaeopigments were negative and gave erratic results, possibly because of low levels, hence only total chlorophyll a values are given without differentiation. Phaeopigment concentrations have been found to be quite low elsewhere in Tasmania, and mostly phaeophytin (pers. comm. Lesley Clementson, CSIRO, Hobart). Mantoura et al., (1997) caution the use of various spectrophotometric methods when degradation pigments are present in either low or high concentrations, and suggest that for accurate determination, or identification, of degradation pigments, TLC or HPLC techniques should be used.

The studies conducted by Hallegraeff et al. (1986), van den Enden (1994) and Brown and McCausland (1999) clearly show the value of assessing phytoplankton species composition and pigment analyses in providing valuable insight into phytoplankton abundance, seston quality, composition of oyster diets and feeding behaviour. Unfortunately, there was insufficient time in the present study to conduct these additional analyses.

3. Hydrodynamics

3.1 Introduction

The hydrodynamic characteristics of shellfish growing areas are important in the assessment of factors which influence oyster growth and conditions within a given region. Knowledge of tidal volumes and flows, exchange rates and residence times provide a means of understanding factors which influence the transport and supply of food material, food quality and quantity and productivity of shellfish growing areas (e.g. Carver and Mallet, 1990; Ball et al., 1997). Estuarine hydrodynamics are complex with flows influenced by many factors, for example, cross-section geometry, residual flows, fresh water inputs, channel depths, extent of shallow water areas, wind generated current flows, friction, storage and barometric pressure (e.g. Kjerfve and Wolaver, 1988; Shetye and Gouveia, 1992; de Jonge, 1992; Cheng et al., 1993; Pejrup et al., 1993; Geyer, 1997). Detailed studies of estuarine currents and hydrology involve quite complex equations and computations.

A simplified approach can be taken based on calculations using information on water volume, bathymetry and tidal heights in an estuary, or embayment (e.g. Williams, 1986; Bell, 1994; Sandford et al., 1992; Luketina, 1998). However, it is noted that calculations based on this information are generally made with the assumption that there is complete mixing of river and seawater, and that there is complete renewal of fresh seawater on the incoming tide (Dyer, 1973; Day, 1981c). As a consequence, estimates of flushing time can be higher due to, for example, incomplete mixing of estuarine water, or return of water from the ebb tide on the flood tide (Dyer, 1973). Sandford et al. (1992) developed a model for estimating tidal flushing of small embayments which uses tidal prism flushing but also incorporates a return flow factor to account for the partial return of water which previously exited on the ebb tide.

As mentioned in the preceding chapter, the three areas studied were representative of different systems broadly classified as estuarine (Little Swanport), marine (Pipeclay Lagoon) and intermediate (Pitt Water). Pitt Water has been classed as intermediate as it is a complex estuarine system where man-made influences have altered the behaviour of the estuary. It was the longest estuary studied, and whilst there is a freshwater input (via

the Coal River) the natural hydrology has been altered following the construction of the Craigbourne Dam in the upper catchment and the weir constructed below the historic Richmond weir. Additionally, a causeway approximately mid-estuary further complicates the hydrology of the estuary. Little Swanport estuary has freshwater inputs with no such similar restrictions, whilst Pipeclay Lagoon is a marine coastal embayment with minimal freshwater inflows. A common feature of each of the areas studied was a narrow constriction at the mouth where tidal exchange occurs, this being almost identical in width at each area.

Data on tidal volumes and flows, exchange rates and residence times for each area were calculated over a spring/neap tide cycle which encompasses high-high, low-low, low-high and high-low tides. These generally occur over an approximate 28 day period (OUCT, 1991).

3.2 Materials and methods

Tide data from each area was measured *in situ* using two types of tide gauges consisting of a pressure sensor encased within a 3 metre length of 50 mm PVC tube secured to a galvanised post driven into the sediment below the approximate low tide mark. At the top of the PVC tube, a waterproof housing was clamped in which the data logger and gel cell battery were located (Fig. 3.1). Tide gauge measurements at Little Swanport were obtained from the type of gauge used in the FRDC study by Crawford et al. (1996). This was a WESDATA (Dataflow Systems™) pressure sensor and data logger with tide heights measured at 1 hour intervals. However, Crawford et al. (1996) noted that these gauges required frequent servicing and many of the 12 units used in that study proved unreliable with little useful information gathered. Of the three gauges positioned in Little Swanport in 1994, one each at sites in the upper, mid and lower estuary, the lower gauge (Fig. 3.2) did yield useful data.

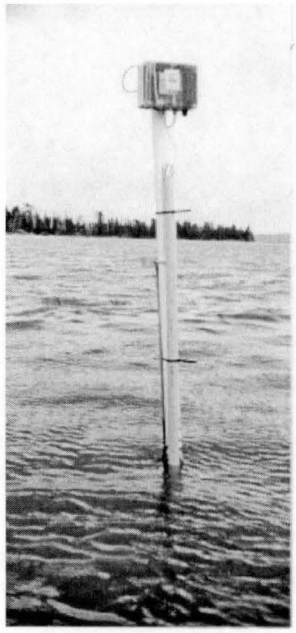

Fig. 3.1 Tide gauge at Lewisham. PVC tubing, with pressure sensor encased within, attached to a galvanised post. Waterproof housing at top in which the data logger and gel cell battery were held.

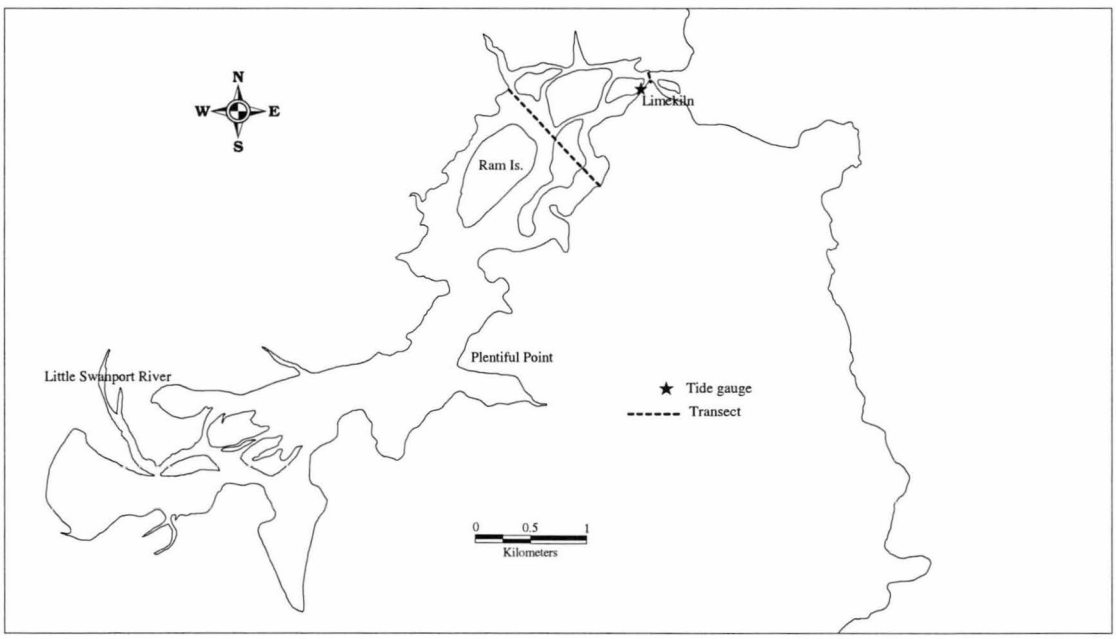

Fig. 3.2 Location of tide gauge and sector transects in Little Swanport.

Three new units were purchased in 1995 to obtain reliable tide data measurements. These tide gauges consisted of a Greenspan™ pressure sensor and Smartreader™ data logger similarly housed in the units described above. All probes were calibrated prior to deployment. The Greenspan gauges recorded tide heights at 30 minute intervals. Three gauges were deployed in Pitt Water during 1995, at Lewisham, south of the causeway and north of the causeway (near the Midway Point Yacht Club) (Fig. 3.3). The tide

gauge at the south of the causeway was relocated to the upper estuary region (Top). In January 1996, the gauges were removed from Pitt Water and relocated to three sites in Pipeclay Lagoon near the mouth of the estuary (Bottom), mid lagoon (Bens Gutter) and in the upper reaches of the lagoon (Top) (Fig. 3.4).

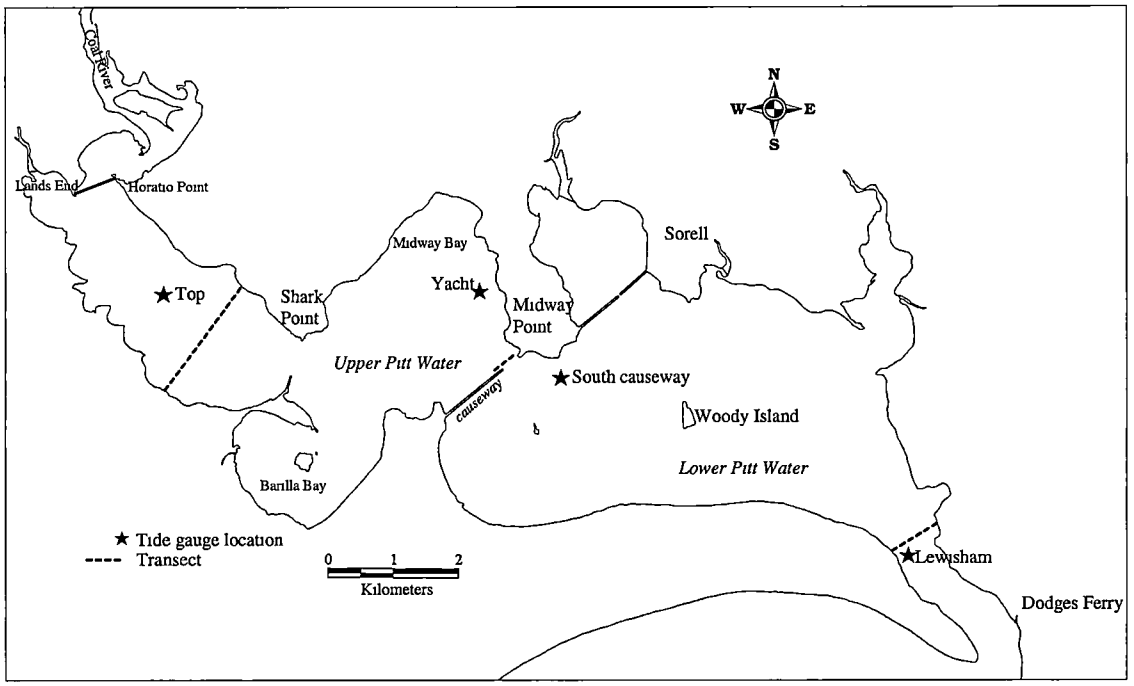

Fig. 3.3 Location of tide gauges and sector transects in Pitt Water estuary. Boundary of the upper estuary from which volumes were calculated (Crawford et al., 1996) is across Lands End to Horatio Point.

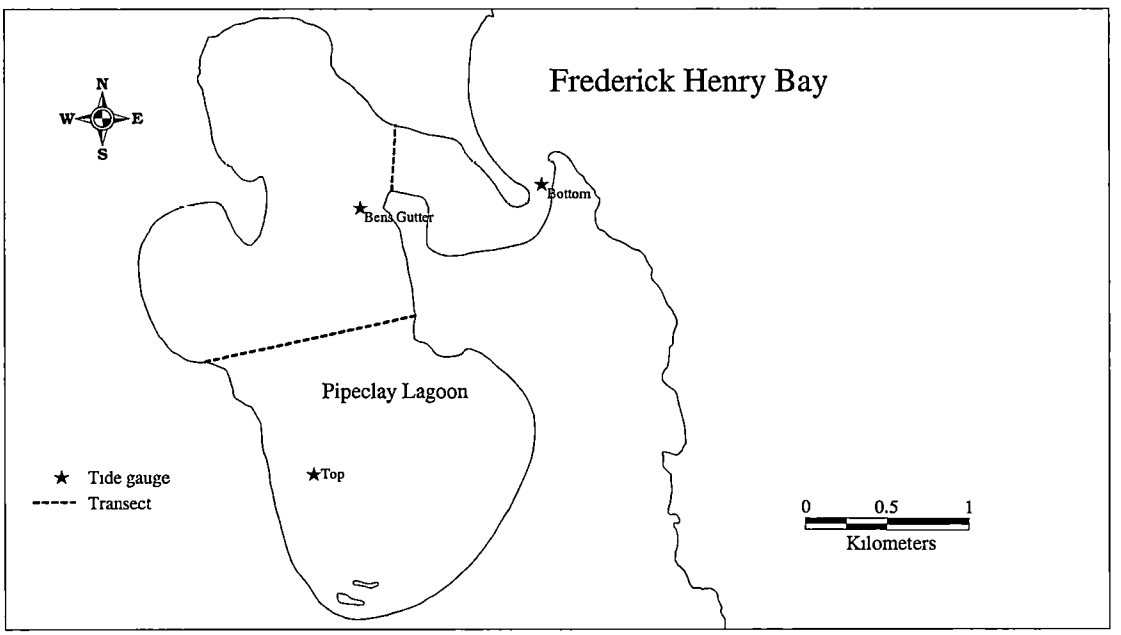

Fig. 3.4 Location of tide gauges and sector transects in Pipeclay Lagoon.

Water volumes and flows were calculated from the tide gauge data and bathymetric information obtained from the study conducted by Crawford et al. (1996). Tidal heights relative to the mean height at each position were calculated by subtraction of the mean height from the raw gauge data. Tidal velocity was calculated from the total volume of water flowing in or out of the body of the lagoon, or estuary, beyond a defined transect (cross-section) for each time interval (30 mins or 1 h) (Fig. 3.5) using the formula below (pers. comm. Dr John Hunter, CSIRO, Hobart).

$$u = \frac{P}{A} \times \frac{\Delta a}{\Delta t} \quad \text{m s}^{-1}$$

where u = velocity m s^{-1}, P is the mean surface area beyond the transect, A is the mean cross-sectional area of the transect, Δa is the change in corrected tide height over Δt the time interval between measurements.

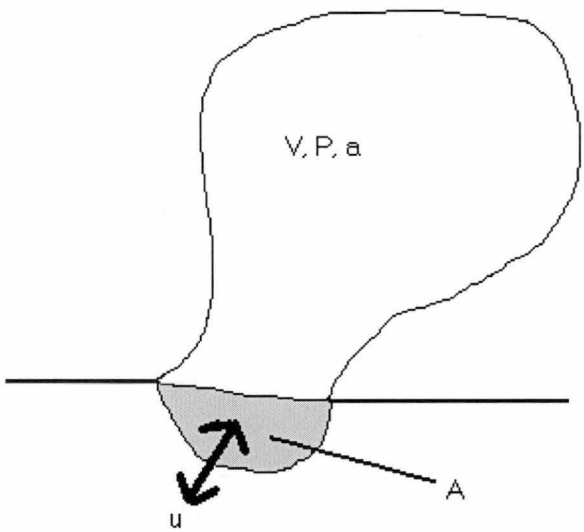

Fig. 3.5 Schematic diagram of estuary/lagoon where V= average volume (m^3), P= mean surface area (m^2), a = change in height due to tide (m), A = mean cross sectional area of transect (m^2) and u = outflow/inflow velocity (m s^{-1}).

Residence time and exchange ratios were calculated over an approximate 28-30 day period using three different methods as follows.

1. $RT = \frac{V + P}{P} \times T$ hours (Luketina, 1998)

where RT is residence time in hours, V = low tide volume (m^3), P = tidal prism (i.e. the difference between high and low water volume), and T = tidal period (i.e. time from high-low-high water (hours)).

2. $RT = \frac{VT}{(1-b)P}$ hours (Sanford et al., 1992)

where RT is the residence time in hours, V = average volume (m^3), T = tidal period (i.e. time from high-low-high water (hours)), b = return flow factor (as detailed following) and P = tidal prism. Sanford et al. (1992) describe the return flow factor b as the fraction of water leaving during the ebb (outgoing) tide which returns on the flood tide. These authors state that in the absence of other means by which to determine b, the setting of b = 0.5 is often recommended. That is, 50% of water leaving on the ebb tide returns on the flood tide. In order to obtain a more accurate figure, detailed studies of coastal water flows would be required to determine the movement of outflow water exiting the mouth of an estuary or lagoon.

3. $r = \frac{P}{V}$ (Williams, 1986)

where r is the exchange ratio, P = tidal prism and V = high tide volume. The flushing time is calculated from the reciprocal of this ratio, i.e. $T = \frac{1}{r}$ (Williams, 1986).

All residence and flushing times are presented as times based on the number of days and part thereof (e.g. 2.6 days). An approximation of the number of tidal cycles can be calculated by multiplying the number of days by 24 hours and dividing by 12.4 (12.4 = 12 hours 25 mins and is the average time of a tidal cycle (OUCT, 1991)).

3.3 Results

3.3.1 Pitt Water

Tide data were recorded at four sites in Pitt Water over the period 16 June 1995 to 16 January 1996 at 30 minute intervals. Mean heights (metre) calculated, number of readings (n), tidal range relative to the mean height and the time periods of recordings at each site were:

Lewisham (1/9/1995 - 19/10/95)	1.031 m (n=2298) (range: - 0.737 to 0.917 m)
South causeway (16/6/95 - 18/8/95)	1.128 m (n=3022) (range: - 0.896 to 0.928 m)
Yacht Club (1/9/95 - 16/1/96)	1.174 m (n=6579) (range: - 0.773 to 0.897 m)
Top (22/8/95 - 16/1/96)	1.090 m (n=7059) (range: - 0.795 to 0.890 m)

Analysis of the tide data for the period 16 June - 30 June 1995 from the gauges positioned north (Yacht Club) and south (South causeway) of the Midway Point causeway bridge showed relatively even tidal heights and similar tidal times across the causeway bridge (Fig. 3.6).

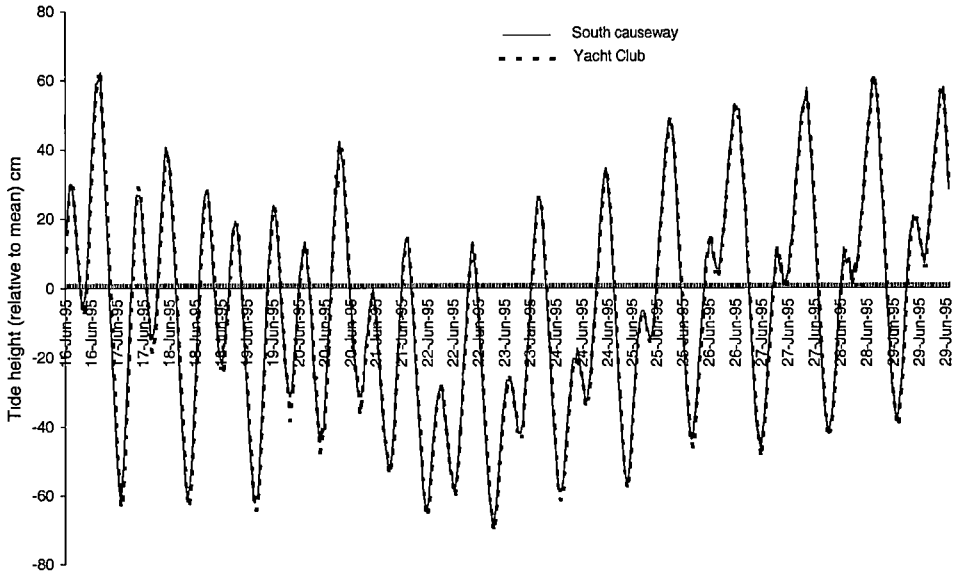

Fig. 3.6 Comparison of tide heights and times across the Midway Point causeway bridge from Yacht Club and South causeway tide gauge for period 16 June to 30 June 1995.

Comparison of tide heights and times along the estuary from the lower estuary region (Lewisham), approximate mid estuary (Yacht Club) and upper estuary (Top) for the period 1 September - 30 September 1995 showed similar tidal amplitude and times for the Top and Yacht Club, with slightly reduced tidal amplitude at Lewisham in comparison. Data recordings at Lewisham also indicated tide times were generally 1 h 29 minutes earlier than those recorded at the upper estuary positions. Representative data for the period 1 to 10 September 1995 is shown below (Fig. 3.7).

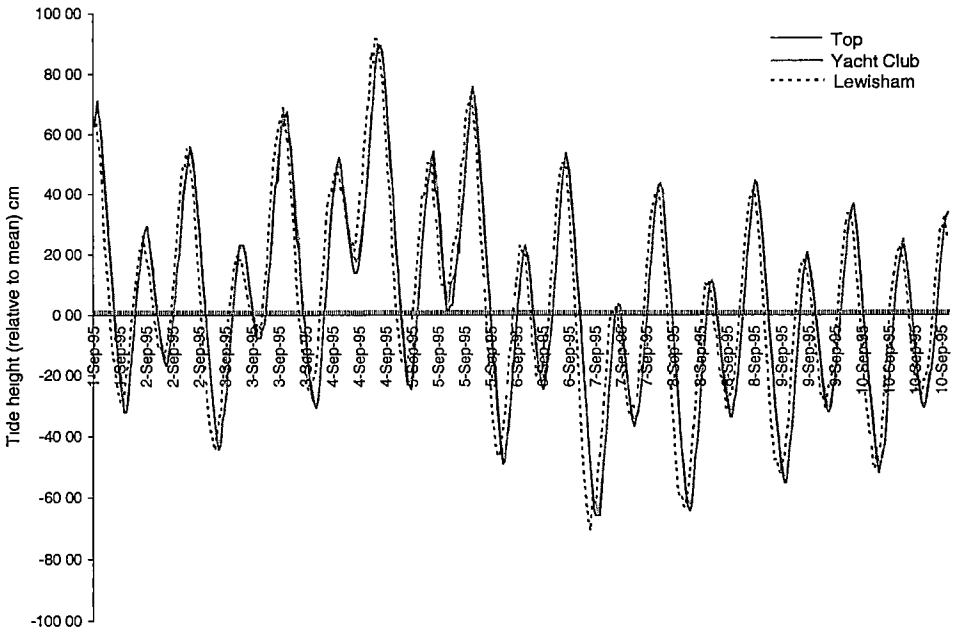

Fig. 3.7. Tidal amplitude and times at lower (Lewisham), approximately mid (Yacht Club) and upper (Top) estuary locations in Pitt Water 1-10 September 1995.

Pitt Water was divided into two box segments representative of the Upper (Box A) and Lower (Box B) estuary, the boundary between the two positioned at the Midway Point causeway bridge (Fig. 3.3). Additionally, for comparison, a transect across the estuary near the Top gauge location in the upper estuary region was analysed to assess approximate tidal flow velocities across the majority of the oyster leases in this region of the estuary. Tidal flows (m s^{-1}) for the period 1-14 September 1995 are shown for comparison (Fig. 3.8). Average surface area, mean cross-sectional area, and mean volumes for Box A, Box B and the Top sector, and average, minimum and maximum ebb and flood velocities calculated for the boundary transects of each box and the upper sector transect for the period 1-29 September 1995 are given in Table 3.1. The surface areas of Upper and Lower Pitt Water are similar, though the Upper region had the greater mean volume of water, approximately 11 million m^3 more than the Lower region.

Table 3.1 Average surface area (km^2), mean cross-sectional area (m^2), mean volume (million m^3), average, minimum and maximum ebb and flood velocities ($m\ s^{-1}$) for Upper, Lower and Top sector of Pitt Water estuary 1-29 September 1995.

	Box A (Upper)	**Box B (Lower)**	**Top Sector**
Surface Area (km^2)	23.06	21.09	6.77
Mean cross-sectional area (m^2)	2692.09	6523.27	4355.4
Mean Volume (million m^3)	41.230	30.154	8.326
Ebb flow - average ($m\ s^{-1}$)	**0.25**	**0.10**	**0.04**
Ebb flow - minimum ($m\ s^{-1}$)	0.00	0.00	0.00
Ebb flow - maximum ($m\ s^{-1}$)	0.80	0.33	0.15
Flood flow - average ($m\ s^{-1}$)	**0.24**	**0.09**	**0.04**
Flood flow - minimum ($m\ s^{-1}$)	0.00	0.00	0.00
Flood flow - maximum ($m\ s^{-1}$)	0.73	0.25	0.13

The greater tidal velocities calculated at the boundary of the Upper and Lower estuary, i.e. the causeway bridge, are due to the narrow constriction at this point of the estuary. Although the causeway is approximately 1.5 km in length, the opening at the bridge is approximately 480 m wide through which a deep channel flows. Turbulent flows are frequently visible in this region during the periods of greatest flow either side of slack water. Reduced current velocities were shown at Lewisham, though these flows were greater than those calculated for the upper estuary region (Top sector). The mean ebb and flood flows at each area were similar in magnitude, however greater maximal flows were calculated on the ebb tides.

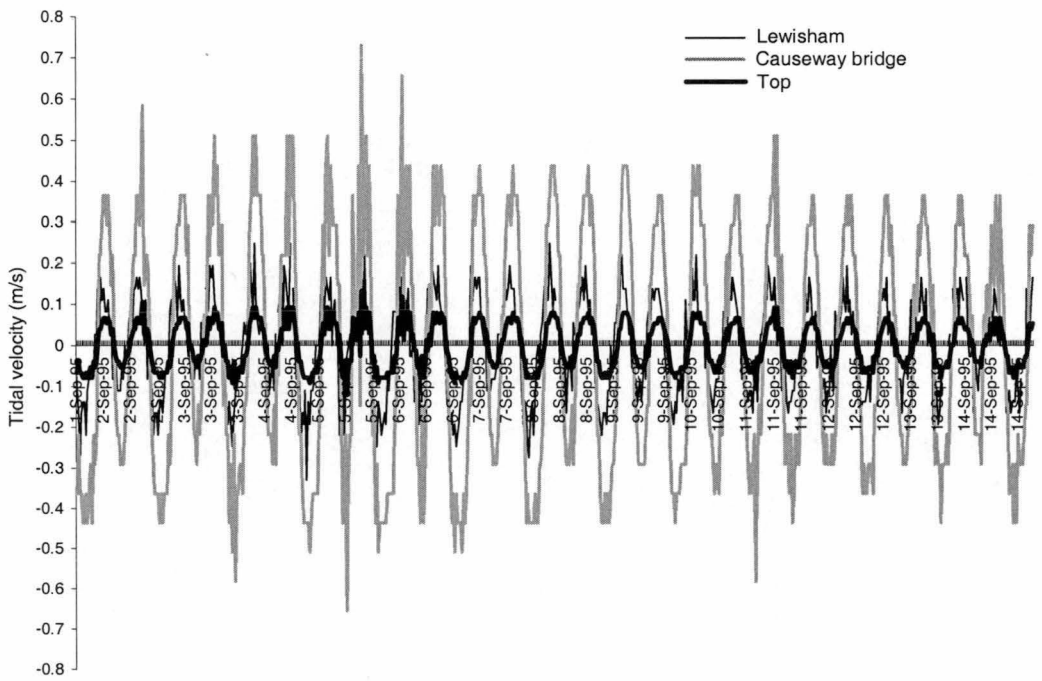

Fig. 3.8 Tidal velocity ($m\ s^{-1}$) in Pitt Water at Lewisham, causeway bridge and Top transect for the period 1-14 September 1995 (30 min intervals). Positive velocities flood (incoming) flows, negative ebb (outgoing) flows.

Average, minimum and maximum total volumes, tidal volumes (prisms) and tidal periods (inclusive of ebb and flood times) were calculated for the Upper (Box A) and Lower (Box B) estuary for the time period 1-29 September 1995 (Table 3.2). The average difference in low tide volume between the Upper and Lower estuary was approximately 10.27 million m^3. Mean ebb tidal prisms were generally greater than the flood tidal prisms, with the greater change in volume occurring in the upper region. Tidal prism in the upper region was approximately 34% of the total volume and in the lower estuary approximately 40% of the total volume in each region. Average tidal period was 12.4 hours, though considerable variation in duration of ebb and flood times was shown. Shorter ebb times, on average, occurred at Lewisham.

Table 3.2 Average, minimum and maximum total volumes, tidal volumes (prisms) and tidal periods (inclusive of ebb and flood times) calculated for the upper (Box A) and lower (Box B) Pitt Water estuary for the time period 1-29 September 1995.

	Average	**Minimum**	**Maximum**
Box A (Upper Estuary)			
Total volume (million m^3)	41.314	25.166	61.904
Ebb Tidal Prism (million m^3)	14.721	4.239	28.613
Flood Tidal Prism (million m^3)	14.438	10.597	18.369
Tidal Period (hours)	12.4	10.0	15.0
Ebb period (hours)	6.2	3.5	8.5
Flood period (hours)	6.3	5.5	7.5
Low tide volume (million m^3)	33.946	25.166	45.654
Box B (Lower Estuary)			
Total volume (million m^3)	30.138	14.611	49.503
Ebb Tidal Prism (million m^3)	12.860	2.261	25.523
Flood Tidal Prism (million m^3)	12.679	8.723	16.800
Tidal Period (hours)	12.4	9.5	15.0
Ebb period (hours)	5.9	3.0	8.5
Flood period (hours)	6.5	5.0	8.0
Low tide volume (million m^3)	23.680	14.611	34.642

Residence times as calculated by the three methods showed some variation in the range of times (Table 3.3). Average residence times were similar between the two regions (range 2-4 days), however longer maximum times were shown in the lower estuary region. Average exchange ratios were similar and relatively low, with greater maximal exchange calculated at Lewisham.

Table 3.3 Average, minimum and maximum residence times (days) calculated using Luketina (1998), Sandford et al. (1992), Williams (1986) and exchange ratio (Williams, 1986) formulae for Upper and Lower Pitt Water 1-30 September 1995.

		Average	Minimum	Maximum
Box A (Upper)	Luketina	2.04	1.15	4.45
	Sandford et al.	3.55	1.68	8.51
	Williams	3.89	2.30	7.94
	Exchange ratio	0.30	0.15	0.46
Box B (Lower)	Luketina	1.85	0.92	6.14
	Sandford et al.	3.17	1.30	11.11
	Williams	3.47	1.81	9.85
	Exchange ratio	0.34	0.15	0.58

3.3.2 Pipeclay Lagoon

Tide data were recorded at three sites in Pipeclay Lagoon over the period 17 January 1996 to 20 May 1996 at 30 minute intervals. Mean heights (metre), number of readings (n) and tidal range relative to the mean height at each site over this time were

Bottom	0.741 m	(n=5950)	(range: - 0.814 to 0.947 m)
Bens Gutter	0.749 m	(n=5948)	(range: - 0.777 to 0.847 m)
Top (corrected)	0.749 m	(n=5948)	(range: - 0.691 to 0.871 m)

The tide gauge positioned at the Top of the lagoon had a mean height of 126.125 cm with mean height corrected for the height of the pressure sensor from the sediment surface (51 cm) as compared to the other gauges. Comparison of tide data at each location over the period 17 January to 15 February 1996 showed similar tidal amplitudes, with an approximate lead time of 45 minutes shown between the Bottom and Top gauges. A representative plot of tide data recorded at each of the three gauges for the period 21-25 January 1996 is shown below (Fig. 3.9). On occasions a lag in tide times was shown between the Top and Bens Gutter gauge during ebb (outflowing) tides.

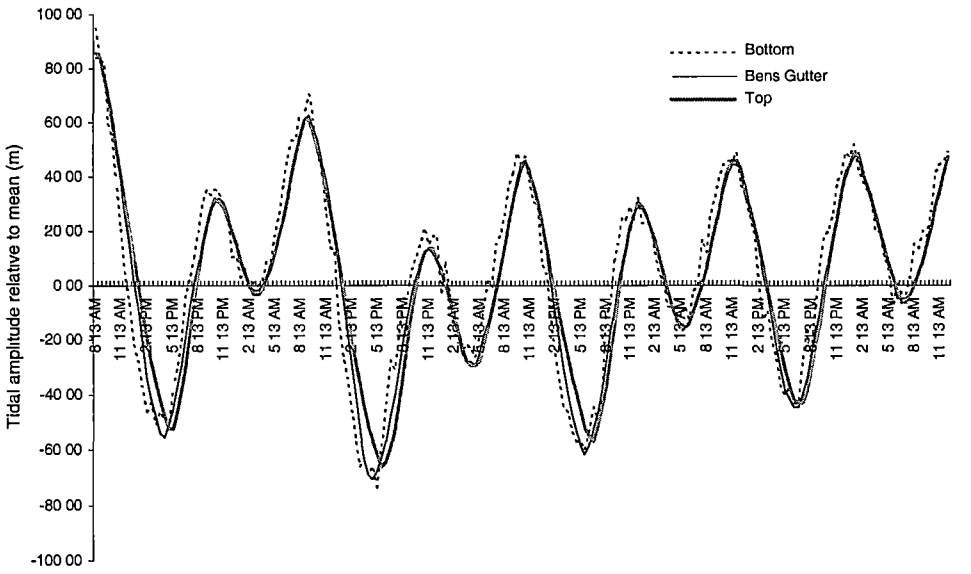

Fig. 3.9 Tidal amplitude and times at lower (Bottom), approximate mid lagoon (Bens Gutter) and upper lagoon (Top) locations in Pipeclay Lagoon 21-25 January 1996.

Tidal velocities were calculated across a transect located at Bens Gutter. Additionally, tidal velocities across a transect located approximately mid-lagoon were calculated to determine approximate flows across the main area where the oyster leases are located (Table 3.4).

Table 3.4 Average surface area (km^2), mean cross-sectional area (m^2), mean volume (million m^3), average, minimum and maximum ebb and flood velocities ($m\ s^{-1}$) for Bens Gutter and mid-lagoon area of Pipeclay Lagoon 17 January - 15 February 1996.

	Bens Gutter	**Mid-lagoon**
Surface Area (km^2)	4.605	2.212
Mean cross-sectional area (m^2)	517.76	857.56
Mean Volume (million m^3)	3.479	1.778
Ebb flow - average ($m\ s^{-1}$)	**0.25**	**0.07**
Ebb flow - minimum ($m\ s^{-1}$)	0.00	0.00
Ebb flow - maximum ($m\ s^{-1}$)	0.83	0.24
Flood flow - average ($m\ s^{-1}$)	**0.24**	**0.07**
Flood flow - minimum ($m\ s^{-1}$)	0.00	0.00
Flood flow - maximum ($m\ s^{-1}$)	0.53	0.15

Greater flows were shown at Bens Gutter, a region where a relatively narrow constriction occurs, as compared to the wider body of the mid-lagoon (Fig. 3.10). Mean ebb and flood velocities were similar in magnitude at each site with flows in the mid-lagoon area approximately 3.5 times slower than those calculated across Bens Gutter.

Generally, maximal ebb flows were greater than flood flows, similarly the higher values occurring at Bens Gutter.

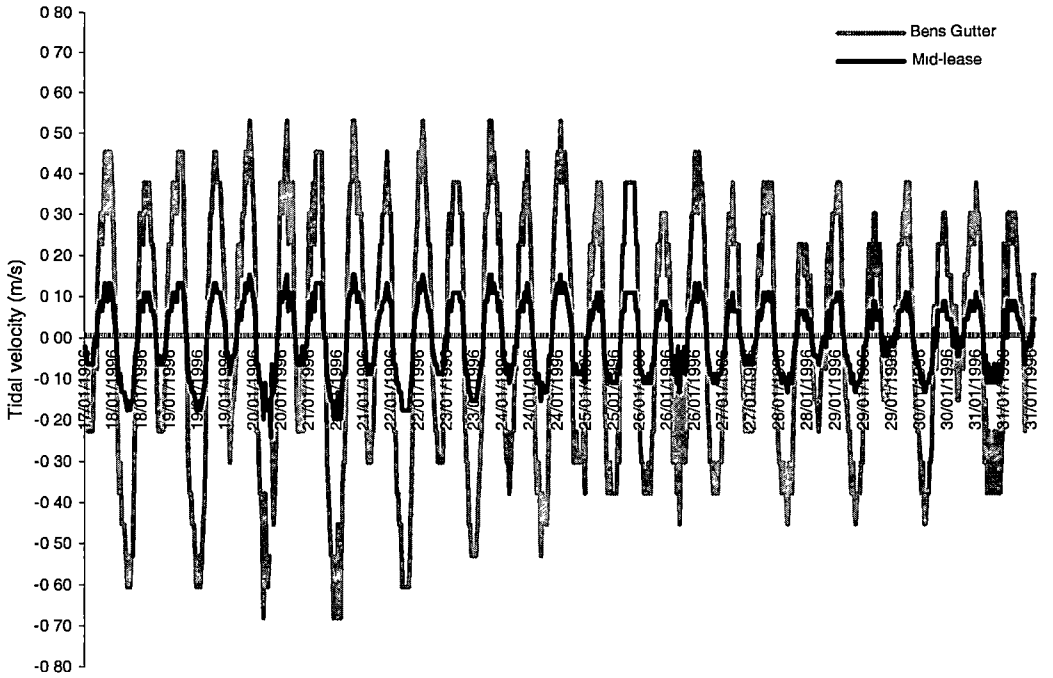

Fig. 3.10 Tidal velocity (m s^{-1}) in Pipeclay Lagoon at Bens Gutter and approximate mid-lease region of the lagoon. Positive velocities flood (incoming) flows, negative velocities ebb (outgoing) flows for the period 17-31 January 1996.

Average, minimum and maximum total volumes, tidal volumes and tidal periods (inclusive of ebb and flood times) were calculated for Bens Gutter over the time period 17 January - 15 February 1996 (Table 3.5). The mean difference between total volume and low water volume was 1.38 million m^3, with the mean tidal prism approximately 73% of the total volume. Ebb and flood tidal prisms on average were similar with ebb flows marginally shorter than flood flows, though some variation is shown in the duration of ebb and flood times.

Table 3.5 Average, minimum and maximum total volumes, tidal volumes (prisms) and tidal periods (inclusive of ebb and flood times) calculated for Bens Gutter 17 January - 15 February 1996.

Bens Gutter:	**Average**	**Minimum**	**Maximum**
Total volume (million m^3)	3.734	0.184	7.379
Ebb Tidal Prism (million m^3)	2.752	0.423	6.419
Flood Tidal Prism (million m^3)	2.791	1.340	4.303
Tidal Period (hours)	12.4	10.0	14.5
Ebb period (hours)	6.0	3.0	8.0
Flood period (hours)	6.4	5.0	7.5
Low tide volume (million m^3)	2.358	0.184	4.557

Residence times as calculated using the three methods were similar with the average residence time of the lagoon approximately 1.5 days (Table 3.6). Maximum residence time was of the order of 5-7 days, with an average exchange ratio of 0.53 and maximum 0.97 which indicates that on occasion there is near complete turnover of the lagoon water within a tidal cycle.

Table 3.6 Average, minimum and maximum residence times (days) calculated using Luketina (1998), Sandford et al. (1992), Williams (1986) and exchange ratio (Williams, 1986) formulae for Bens Gutter, Pipeclay Lagoon 17 January - 15 February 1996.

		Average	Minimum	Maximum
Bens Gutter	Luketina	1.37	0.60	4.90
	Sandford et al.	2.05	0.65	6.85
	Williams	1.24	0.53	6.08
	Exchange ratio	0.53	0.09	0.97

3.3.3 Little Swanport

Tide data were recorded at Limekiln for a period of approximately 12 months in 1994, however, only the January data was used for this study. Tide heights and times were recorded at 1 hour intervals using a different tide gauge system to that used at Pitt Water and Pipeclay. Tidal heights for the period 1-29 January 1994 as recorded at Limekiln, near the mouth of the Little Swanport estuary are shown below (Fig. 3.11).

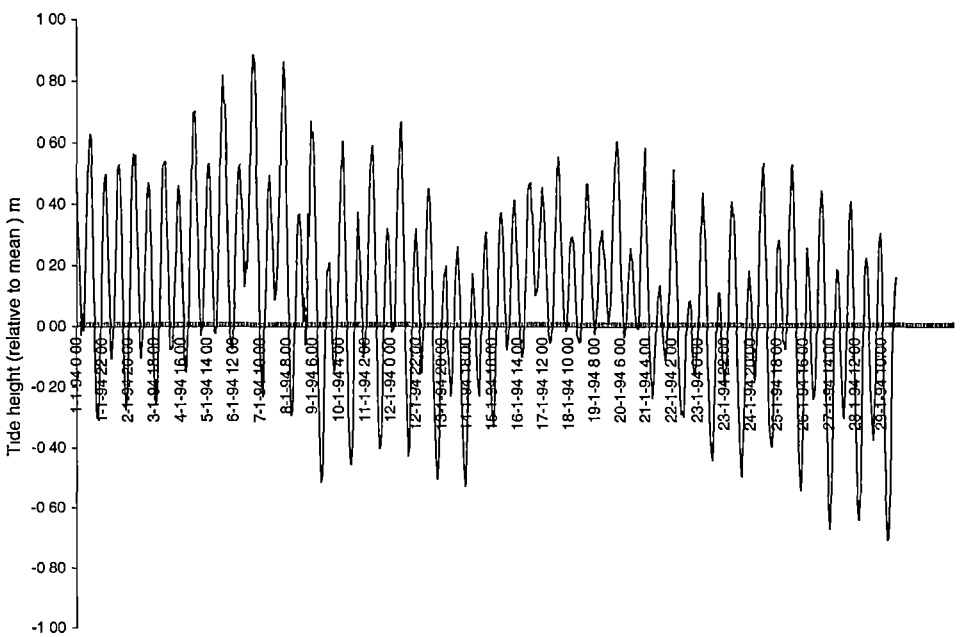

Fig. 3.11 Little Swanport tidal amplitude for the period 1-29 January 1994 as recorded at Limekiln Point (near the mouth of the estuary).

Tidal velocities were calculated across a transect located near the mouth of the estuary at Limekiln. Additionally, tidal velocities across a transect located approximately mid-lease area of the lower estuary region were calculated to determine approximate flows across the main area where the oyster leases are located (Table 3.7).

Table 3.7 Average surface area (km^2), mean cross-sectional area (m^2), mean volume (million m^3), average, minimum and maximum ebb and flood velocities ($m\ s^{-1}$) at Limekiln and mid-lease area of the lower estuary of Little Swanport, 1-29 January 1994.

	Limekiln	**Mid-lease area**
Surface Area (km^2)	6.321	5.532
Mean cross-sectional area (m^2)	315.0	1171.0
Mean Volume (million m^3)	6.554	5.725
Ebb flow - average ($m\ s^{-1}$)	**0.61**	**0.14**
Ebb flow - minimum ($m\ s^{-1}$)	0.00	0.00
Ebb flow - maximum ($m\ s^{-1}$)	1.73	0.41
Flood flow - average ($m\ s^{-1}$)	**0.57**	**0.13**
Flood flow - minimum ($m\ s^{-1}$)	0.00	0.00
Flood flow - maximum ($m\ s^{-1}$)	2.07	0.49

The greater mean velocities were calculated near the mouth of the estuary (0.61 $m\ s^{-1}$), where there is a narrow constriction, as compared to the broader area of the leases in the lower estuary region (0.14 $m\ s^{-1}$), the differences due to the different cross-sectional area of each region. Mean velocities near the mouth were approximately four times faster than those across the lease area in the lower estuary. On average, ebb flows were marginally greater than flood flows, however, mean maximal flows were greater during flood tides than ebb. Tidal velocities for Limekiln and the transect across the mid-lease region of the lower estuary are shown below (Fig. 3.12).

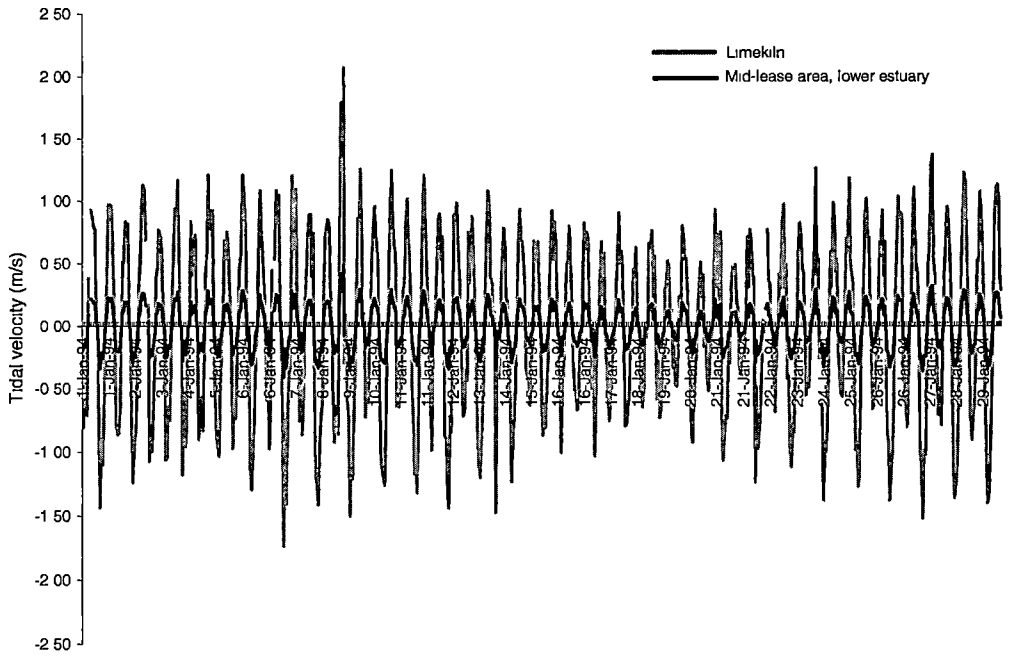

Fig. 3.12 Tidal velocity (m s^{-1}) near mouth and mid-lease area of the lower estuary of Little Swanport 1-29 January 1994. Positive velocities flood (incoming) flows, negative velocities ebb (outgoing) flows.

Average, minimum and maximum total volumes, tidal volumes and tidal periods (inclusive of ebb and flood times) were calculated for Limekiln over the time period 1-29 January 1994 (Table 3.8). The mean difference between total volume and low water volume was 2.04 million m^3, with the mean tidal prism approximately 55% of the total volume. Ebb and flood tidal prisms on average were similar, with ebb flow duration marginally shorter than flood flows, though some variation is shown in the duration of ebb and flood times.

Table 3.8 Average, minimum and maximum total volumes, tidal volumes (prisms) and tidal periods (inclusive of ebb and flood times) calculated for Limekiln (Little Swanport) 1-29 January 1994.

Limekiln:	**Average**	**Minimum**	**Maximum**
Total volume (million m^3)	7.300	2.335	12.148
Ebb Tidal Prism (million m^3)	4.052	1.539	7.473
Flood Tidal Prism (million m^3)	4.033	1.834	5.457
Tidal Period (hours)	12.4	9.0	16.0
Ebb period (hours)	6.0	4.0	9.0
Flood period (hours)	6.3	4.0	8.0
Low tide volume (million m^3)	5.263	2.335	7.399

Residence times as calculated using the three methods were similar, with the average residence time of the estuary approximately 1.6 days (Table 3.9). Maximum residence time was of the order of 2.4 - 4.3 days, with an average exchange ratio of 0.43 and

maximum 0.77, which indicates that on occasion there is a reasonable turnover of the estuary water.

Table 3.9 Average, minimum and maximum residence times (days) calculated using Luketina (1998), Sandford et al. (1992), Williams (1986) and exchange ratio (Williams, 1986) formulae for Limekiln, Little Swanport 1 -29 January 1994.

		Average	Minimum	Maximum
Limekiln	Luketina	1.35	0.72	2.66
	Sandford et al.	1.97	1.01	4.26
	Williams	1.33	0.72	2.42
	Exchange ratio	0.43	0.21	0.72

3.4 Discussion

Pitt Water was the largest of the regions studied, having a total surface area approximately 10 times greater than Pipeclay Lagoon and 7 times greater than Little Swanport estuary. Slower flows were calculated in the upper region of the estuary as compared to those at Lewisham, with rapid tidal flows occurring at the causeway bridge due to the narrow constriction at this point. The higher mean tidal heights recorded on either side of the causeway bridge (i.e. Yacht and South causeway), most likely could be due to the effects of the constriction to flows at this point. Similar observations of elevated tidal heights at points of constriction were recorded by Geyer (1997), attributed to the banking-up of the water at the point of constriction.

The mean volume of Upper Pitt Water was approximately 11 million m^3 greater than Lower Pitt Water, due to the greater amount of deeper water in this region, particularly in the area north of the causeway, compared to the more extensive shallow water areas in the lower estuary, as shown from the bathymetric information of the estuary (Crawford et al., 1996).

Greater flow rates were calculated on ebb tides, though considerable variation was shown in the duration of tide flow times. Variation in tide times and heights have been shown to be influenced by barometric pressure and strength and direction of prevailing winds (e.g. OUCT, 1991; de Jonge, 1992). Similarly, ebb tidal prisms were greater than flood, with the ebb tidal prisms less at Lewisham than the upper region. Given the difference in tide times and calculated flow rates, it appears that outflowing water from the upper region reaches the lower estuary at the time of incoming water. The shorter ebb times indicated at Lewisham appear to support this, thus residual flows from the

upper estuary most likely contribute to the increasing tidal heights and earlier tide times of the lower estuary.

Lewisham is an interface site of the meeting of outflowing with inflowing water (pers. obs.). Harris (1968) noted sites of mixing zones within the estuary at Lewisham, southwest of Woody Island, the causeway bridge and at Shark Point, these locations being points where residual ebb tidal currents meet flood tidal currents. These interference points are marked by strong turbulence and whirlpools in the water (Harris, 1968). Pitt Water is characterised by a deeper relatively narrow channel which stems from the inlet at Dodges Ferry to Woody Island ending in, what Harris (1968) describes as, a fan of diverging channels. A clearly defined channel resumes south of the causeway bridge and continues through the upper estuary region. Additionally, Harris (1968) noted separate ebb and flood channels, such as the fan at Woody Island, except at points where currents were restricted to a single channel, for example Shark Point and the causeway bridge. These bedform features reflect the dominant hydrological patterns within the estuary and provide a greater understanding of ebb and flood tidal flow characteristics within the estuary.

Calculated tidal prisms approximated 34-40% of the total volume, with a mean exchange ratio of 0.3. Residence time of the upper estuary was calculated to be within the range of 2-4 days, however, if considering the estuary as a whole, a more appropriate return flow factor (as per Sandford et al. (1992)) would be of the order of b = 0.25, which means a more appropriate approximation of residence times in the upper estuary is 7 - 17 days.

Pipeclay Lagoon showed a greater exchange of marine water, due to its smaller size by comparison to Pitt Water. The mean tidal prism approximated 73% of the total volume, and suggests that a large proportion of the lagoon water volume is replaced by tidal flows and reinforces the marine nature of this lagoon system. The average residence time was calculated to be approximately 1.5 days with an average exchange ratio of 0.53 and maximum of 0.97 (i.e. near complete turnover of the lagoon water on occasion). Flows across the mid-lease area of the lagoon were approximately 3.5 times slower than those at Bens Gutter. However, tidal flows calculated were faster than those shown across the lease area of Pitt Water (mean 0.07 m s^{-1} at Pipeclay compared to 0.04 m s^{-1} at Pitt Water). Similarly to Pitt Water, ebb flows on average were faster than flood flows.

Mean flows across the lease area of the mid-lower estuary region of Little Swanport were much greater than those calculated at Pitt Water and Pipeclay Lagoon (mean 0.14 m s^{-1}). Tidal flows across the mouth of the estuary were also greater than those calculated at the other areas, with maximal flows of 2.07 m s^{-1}. The average tidal prism was approximately 55% of the total volume of the estuary water with a mean exchange ratio of 0.43 and maximum of 0.72. The average residence time was calculated to be approximately 2 days with a maximum of 4.3 days. Little Swanport is characterised by a relatively deep channel within the mid-estuary region which extends from the south-eastern side of Ram Island to near the upper sample site location (Dyke). This deep channel most likely is a point of mixing of riverine water with marine waters and the greater depths have probably been created by scouring, due to the relatively narrow constriction within this region of the estuary as well as turbulent flows during mixing.

On average, ebb flows were stronger and of shorter duration than flood flows in the three areas (i.e. Pitt Water, Pipeclay Lagoon and Little Swanport) and therefore they could be described as being ebb-dominant systems (Shetye and Gouveia, 1992). Such systems are characterised by a deep sub-tidal channel with extensive areas of mudflats (or shallow water regions) (Shetye and Gouveia, 1992). The range in ebb and flood flows and tidal prisms (i.e. minimum and maximum values) are attributed to variations in tidal heights, from minimal differences between high and low water to maximal differences over the representative 28-30 day period used in the calculations at each area.

The theoretical calculations of residence times and exchange ratios tend to support the observations noted in each area, following the period of heavy rainfalls and flood events in late 1995 to early 1996. The longer residence times of Pitt Water are reflected in the longer recovery time of the estuary to more normal salinities following flooding of the estuary. Since the construction of the Craigbourne Dam, such events are less frequent than historical patterns, with a study by Daley (1999) finding that flow response of the Coal River to rainfall events has diminished. Daley (1999) indicated that post-dam, catchments below the Craigbourne Dam contribute to flood flows with only a small proportion of catchment flow reaching the lower reaches of the estuary. The response of river flow to rainfall and flooding of the lower estuary, as indicated by reduced salinity, is shown (Fig. 3.13). The cumulative effects of the four peak discharge events during December to early February considerably reduced the salinity of the estuary with a gradual increase in salinity shown at the last sampling.

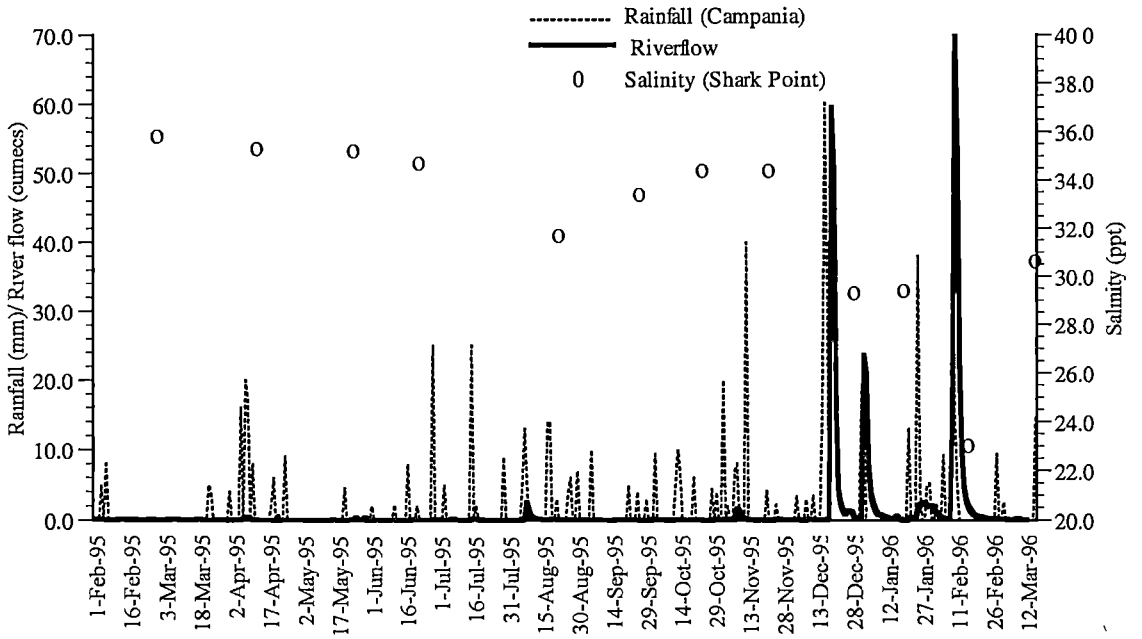

Fig. 3.13 Rainfall, riverflow and salinity (Shark Point) for Pitt Water February 1995 to March 1996. (source riverflow data DPIWE, Land and Water Resources Division - Coal River at Richmond flow gauge).

Similarly, salinities at Little Swanport were depressed during this same time period (refer Chapter 2), as a consequence of heavy and prolonged rainfalls. However, no riverflow data are available by which to make a similar comparison, due to the closure of the flow gauge on the Little Swanport River. Sampling, however, indicated a comparatively more rapid return to normal salinities than those shown at Pitt Water. This most likely is attributed to the higher exchange ratio and shorter residence time calculated for this estuary. Pipeclay Lagoon showed minimal reductions in salinities during the period of heavy rainfalls during January-February 1996. This was attributed to the more marine nature of this lagoon and rapid flushing times.

Of the three methods used in this study to calculate residence times, the method of Sandford et al. (1992) is considered the more appropriate, as it accounts for the return of water previously resident, and hence is more realistic of conditions which occur in the areas studied. Ideally, more rigorous calculations using the methods of Luketina (1998) and Williams (1986) should be based on greater segmentation of the estuary, or embayment, with flushing time calculated from the sum of the segment flushing times.

4. Oyster biodeposition

4.1 Introduction

In recent times, there has been considerable expansion of the Tasmanian aquaculture industry with further planned increase in farming areas. With this development of marine farms around the state of Tasmania, concern has been raised by the community with respect to the culture of shellfish. One issue frequently raised is the amount of biodeposition and the potential detrimental environmental impact this may have. Figures from overseas have often been cited in objections to marine farm applications (pers. comm. Margaret Brett, DPIWE, Marine Farming Branch) because of the paucity of information from Tasmania, or Australia.

Considerable studies have been conducted overseas to assess the impact of fin fish aquaculture, where there is the added complication of external loading via the addition of fish food (e.g. Woodward, 1989; Wu et al., 1994; Sowles et al., 1994; Tsutsumi, 1995; Cheshire et al., 1996; Karakassis et al., 1998). However, continued efforts are being made to reduce the degree of these impacts by appropriate site selection, better feeding systems (feed loss is expensive), appropriate stocking rates, rotational cage/farm practices (or fallowing), improved technology to capture wastes from cages ('bag' systems) and stricter environmental monitoring requirements (e.g. Silvert, 1992; Rosenthal, 1994b; Black and Truscott, 1994; Ang and Petrell, 1997; Ervik et al., 1997). By comparison, few studies have been conducted on shellfish culture, though generally studies on oyster and mussel culture have found impacts to be less than those associated with fin fish culture (Dahlbäck and Gunnarsson, 1981; Sornin et al., 1983; Kaspar et al., 1985; Boucher and Boucher–Rodoni, 1988; Hatcher et al., 1994; Grant et al., 1995; Villareal, 1995; Barranguet, 1997; Casabianca et al., 1997; Kaiser et al., 1998). A mitigating factor with shellfish culture is that it does not have the added complication of external feed inputs, as shellfish are reliant on food available naturally in the water column.

In filtering particles from the water column, material is either rejected and expelled as pseudofaeces, or ingested and later expelled as faeces. It is this material, collectively known as biodeposits (faeces and pseudofaeces combined), which is deposited to the

sediments. The degree of biodeposition is largely a function of the scale of culture operation: obviously, dense stocks of shellfish will result in greater loading of biodeposits to the substrate per unit area. The type and amount of material deposited is influenced by the ambient quantity and quality of seston (Iglesias et al., 1998). Studies have shown that selectivity of particulate matter by filter feeders, predominantly occurs prior to ingestion (Kiørboe and Møhlenberg, 1981; Shumway et al., 1985; Lucas et al., 1987; Barillé et al., 1993; Pastoureaud et al., 1996; Navarro and Thompson, 1997; Hawkins et al., 1998). Greater rates of pseudofaeces production have been shown, for example, to be related to high seston loads, when seston is composed of a large fraction of material 'outside' the size range capable of being ingested, or when seston consists of a high inorganic or 'undesirable' component (Sornin et al., 1988; Barillé and Prou, 1993; Pastoureaud et al., 1996; Dame, 1996; Hawkins et al., 1998).

The sedimentation of biodeposits to the substrate and subsequent degradation, or remineralization, is not only influenced by the rate of supply, but also by the quality of the material deposited. Shellfish increase the natural sedimentation rate of particulate material from the water column to the sediments by filtration and ingestion, with subsequent alteration to the composition and size spectrum of the particulate matter (Haven and Morales–Alamo, 1966; Bernard, 1974; Kautsky and Evans, 1987; Dame et al., 1991). Pseudofaeces, which consist of rejected material bound in a mucus substance (Kiørboe and Møhlenberg, 1981), are generally lighter than faeces and hence take longer to settle out. They can also be transported further in the water column. The composition of pseudofaeces is variable and is dependent on the material rejected. Studies have shown a much greater inorganic fraction in pseudofaeces, due to preferential uptake of organic particles (e.g. Hawkins et al., 1997; Hawkins et al., 1998). In contrast, Navarro and Thompson (1997) showed faeces to contain a higher content of inorganic material than the seston or pseudofaeces. High levels of chlorophyll a were found in the pseudofaeces, which these authors attributed to removal of excess phytoplankton during bloom conditions. The rate of mineralization of faeces has been shown to be greater than pseudofaeces and it has been suggested that this is due to higher loads of bacteria excreted along with faeces (Smaal and Prins, 1993). In the process of filtration and ingestion, the possibility exists for ingestion and concentration of bacteria, either free or attached to particles selected. On ejection of faeces these localised concentrations of bacteria can readily utilise nutrients within the faeces. Lucas et al. (1987) showed high levels of bacterioplankton clearance and indicated that while

phytoplankton were a preferred nutritional source, in regions where bacterial biomass was high relative to other particulate sources, these could provide an important source of carbon and nitrogen to bivalve filter feeders.

Loading to the sediment surface is also dependent on water depth and current flows. Greater rate of dispersion is possible, when water depths are deeper and current flows high or in shallow regions with fast current flows (predominantly tidal currents). Under these conditions, greater transport of biodeposits in the water column occurs before settlement to the substrate. Depositional loadings of fish faeces and fish feed under fish cages have been calculated to determine the rate of loading of this material to the substrate, based on predicted particle settling rate (fish feed or faeces), current flow and water depth (Woodward et al., 1992; Gowen et al., 1994; Silvert and Sowles, 1996; Hevia et al., 1996). Another consideration, especially for shallow water regions, is wind driven resuspension of sediments and hence dispersion of biodeposits.

The rate of degradation of biodeposits upon reaching the sediment surface, is dependent on factors such as temperature, oxygen availability and the benthic macro and micro community. Biodeposits can enhance the benthos, by the supply of organic rich material (Bernard, 1974; Kautsky and Evans, 1987; Sornin et al., 1990; Hatcher et al., 1994; Barranguet et al., 1994; Grant et al., 1995). Stimulation of microbial biomass as a consequence of this, in addition to supply of bacteria via faeces (as described above), would also enhance rapid degradation of this organic material. However, excessive loadings beyond the assimilatory capacity of the benthos can lead to accumulation of organic matter resulting in detrimental conditions of anoxia and changes in the benthic assemblages, or in extreme cases azoic conditions.

Changes in the macrobenthic community have been shown directly under oyster culture installations at sites in Mexico as a consequence of oyster biodeposition (Villarreal, 1995). These locations showed a marked change in community composition, signs of eutrophication, as well as reduction in seagrass (*Zostera marina*) density. However, Villarreal (1995) reported that areas where oysters had been removed were recolonised by *Zostera marina* within four months, though the invertebrate community did not complete recolonisation equivalent to unaltered areas of *Zostera marina* meadows until approximately six months later. Everett et al. (1995) noted a rapid decline in eelgrass (*Zostera marina*) cover after 9 months following installation of oyster (*Crassostrea gigas*) racks to almost complete absence of eelgrass at 18 months. However, loss of

cover at 18 months was confined to a region approximately 2 m either side of the centre of the racking with relatively dense cover measured beyond this distance (Everett et al., 1995). Everett et al. (1995) attributed the loss of eelgrass within this region of racks to sediment erosion and possibly shading. A study of native mussel beds (*Mytilus edulis*) in Kiel Fjord (Western Baltic) by Reusch et al. (1994) showed that natural beds of mussels enhanced eelgrass (*Zostera marina*) growth from the biodeposition of nutrient-rich material by the mussels. Subsequent mineralisation of this material increased the amounts of nutrients (nitrogen in particular) available to the eelgrass rhizosphere.

Dahlbäck and Gunnarsson (1981), in a study under long line mussel culture in Sweden, noted significant effects due to biodeposition. These authors reported soft, black sediments (with strong hydrogen sulphide odour), 40-50% *Beggiatoa* cover, elevated %OM content and increased sulphate reduction due to enhanced bacterial activity under the mussel culture as compared to reference sites outside. However, it should be noted that this study was conducted in an area where the tidal amplitude was minimal (0.3 m) and current flows were weak (mean 3 cm s^{-1}). Similarly, Grant et al. (1995) noted elevated anaerobic activity in the sediments under long line mussel culture at a site in Nova Scotia, though the appearance of *Beggiatoa* was sporadic and only apparent in the summer of their study. The most notable feature of the study by these author's was that the benthic macrofauna was more abundant at the reference site, but biomass was generally lower than at the mussel site. The mussel site also had a predominance of mollusc species. It was concluded that the impact of fallen mussels to the sediment, and hence influence of community structure in response to this (predation), was more significant than the impact due to organic enrichment or hypoxia. Unlike the study site of Dahlbäck and Gunnarsson (1981), current flows at Grant et al. (1995) study site were greater, with peak flows of 15 cm s^{-1} reported. Both of the above studies were of sub–tidal cultivations. De Grave et al. (1998) examined changes in benthic macrofauna within an intertidal oyster (*Crassostrea gigas*) culture site. These authors found no evidence of organic enrichment under cultivation sites as compared to reference sites, with greater changes shown in the benthic community structure of the access lanes (corridors used by farm service vehicles) as compared to under culture sites. This was attributed to compaction of the sediments as a result of heavy vehicle traffic.

A number of studies have been conducted on biodeposition principally to determine 'food' assimilation rates, where assessment was made based on the composition of inflow and outflow water. Some studies however, were concerned with the quantity and

composition of biodeposits and how these related to variation in the seston, or natural particulate sedimentation (Haven and Morales–Alamo, 1966; Bernard, 1974; Sornin et al., 1983; Kautsky and Evans, 1987). Biodeposits produced were collected and analysed and, from comparison to the inflow and outflow water composition, the assimilation of material by the shellfish determined. Many of these studies were performed on a small number of isolated animals held within experimental enclosures (special purpose containers, trays or raceways) (Haven and Morales–Alamo, 1966; Foster–Smith, 1975; Valenti and Epifanio, 1981; Bayne et al., 1987; Sornin et al., 1988; Bayne et al., 1989; Carver and Mallet, 1990; Barille and Prou, 1993), holding a small number of animals on purpose built sediment traps *in situ* (Kautsky and Evans, 1987), or *in situ* traps placed under culture structures (Sornin et al., 1983). Animals were either subjected to diets of known composition (eg mono– or mixed algal species, additions of silt) or ambient water, with extrapolation of results to the population, or field, conditions.

Assessment of sedimentation due to biodeposition has also been determined from the measurement of topographical changes in sediment height, due to biodeposition or the effect of culture structures (Everett et al., 1995; Hone, 1996). Generally, these have involved two procedures, one which measures the height of the sediment profile from a horizontal reference 'string' line, or sediment build up on a plate placed in the sediment. The latter procedure, however, may be affected by disturbing and disrupting the integrity of the sediment, when the plate is put in place. Hone (1996) noted that both of the methods mentioned above failed to measure any increase or decrease in sedimentation associated with oyster leases in South Australia. This was attributed to the low pseudofaeces production by the oysters, due to the low ambient sediment content. However, it was noted that current flow within the lease was reduced, due to rack structures (Hone, 1996), though no mention was made of faecal production or sedimentation of this material. In contrast, Everett et al. (1995), employing a similar technique of horizontal reference line, noted sediment erosion around rack structures with sediment directly under the rack structures slightly lower than that immediately adjacent to the racks. Sediment grain size under the racks was found to be considerably coarser than at reference points outside the racking, indicating scouring effects (Everett et al., 1995). However, unlike the observations noted by Hone (1996) in South Australia, the study by Everett et al. (1995) was conducted in an area having a large tidal range of approximately 4 m and fast average current flows of the order of 0.5 m s^{-1}.

A number of aspects need to be considered with the use of sediment traps and the design utilised (Hargrave and Burns, 1979: Gardner, 1980a; Gardner, 1980b; Lorenzen et al., 1981; Håkanson et al., 1989). The shape and type of sediment traps are important, for example with cylindrical traps the height to width ratio must be selected, and the use of baffles to minimise eddying effects considered. Suitability of design and trap efficiency must be weighed against the purpose and conditions of the study, with acceptance and recognition of any limitations which may ensue. With this study, a similar approach to that used by Sornin et al. (1983) was used, where large funnels with collection jars attached were used to collect material from under oyster culture baskets and at control sites.

In situ studies of biodeposition rates of Pacific oysters (*Crassostrea gigas*) were conducted on an intertidal shellfish lease at Pipeclay Lagoon on 25-28th November 1995 and again 15-16th June 1996. The objectives of this study were to determine rates of biodeposition, the quality of this material and assessment of biodepositional loading under culture installations. No such studies have been conducted in Tasmania, or Australia.

4.2 Materials and methods

4.2.1 Biodeposition

An *in situ* study of biodeposition rates of Pacific oysters (*Crassostrea gigas*) was conducted on a 10.3 ha intertidal shellfish lease located in Pipeclay Lagoon (Fig. 4.1). Sampling was conducted on two occasions, summer (November 1995) and winter (June 1996). Sampling was undertaken on days when no work was done on the lease (Friday to Sunday), to avoid possible influences from farm operational activities (e.g. boat traffic, movement of stock). The lease area was divided into 20 m grids and at each sampling period, random numbers were generated to select the location of trap sites.

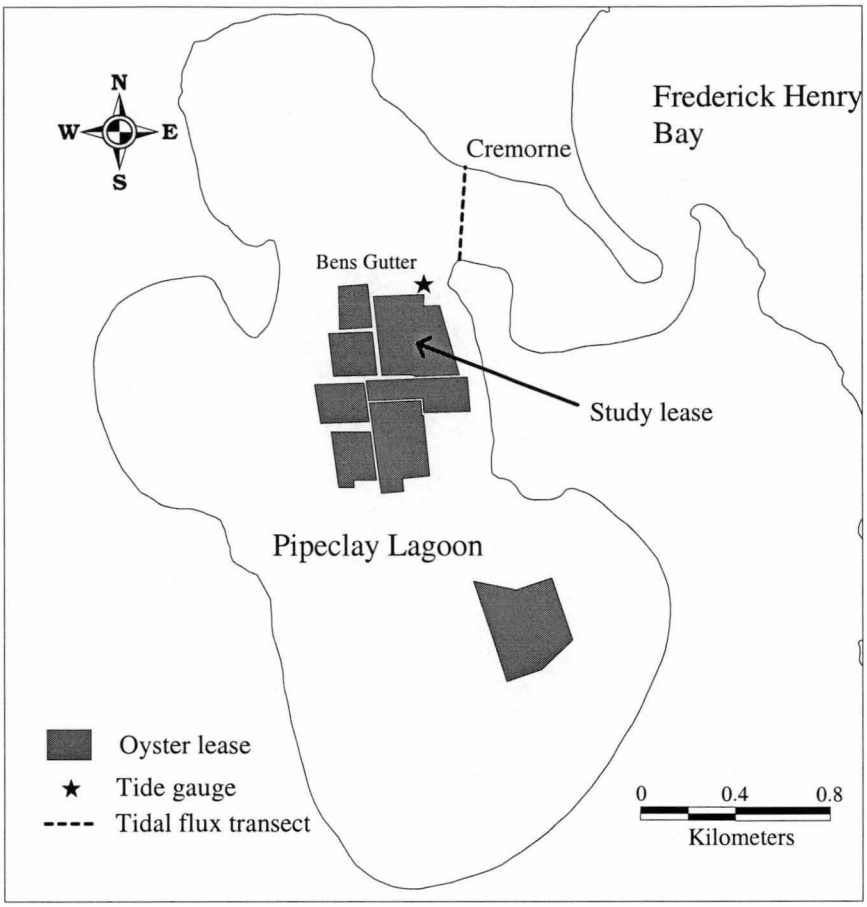

Fig. 4.1 Location of oyster biodeposition study lease, tide gauge and tidal flux transect in Pipeclay Lagoon.

Sediment traps similar to the design and approach used by Sornin et al. (1983) were used in this study. Traps consisted of a plastic 35 cm diameter funnel with a coarse mesh screen near the base of the funnel and a 500 ml screw cap jar (with a hole cut in the screw cap) attached to the bottom (Fig. 4.2). Total height of the traps was 35 cm. Surface area of the mouth of the trap (funnel) was 0.0962 m^2. Rack height within the study lease was variable from approximately 0.2 to 0.8 m high, generally 0.5 to 0.7 m.

A total of twelve traps were deployed, eight traps set under oyster baskets (Fig. 4.3), and four on racking without oysters (controls) (Fig. 4.4). During each period, the traps remained in place and the collection jars were changed daily at low water.

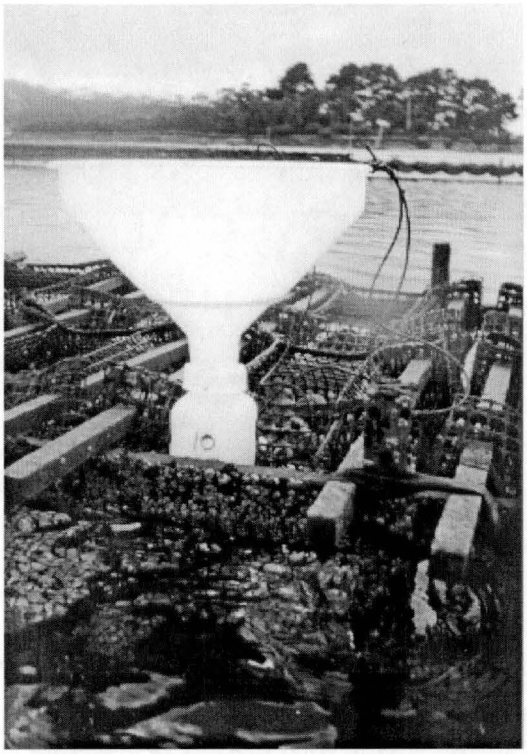

Fig. 4.2 Sediment trap constructed from 35 cm diameter funnel and 500 ml collection jar. Oyster culture baskets and arrangement on racking shown.

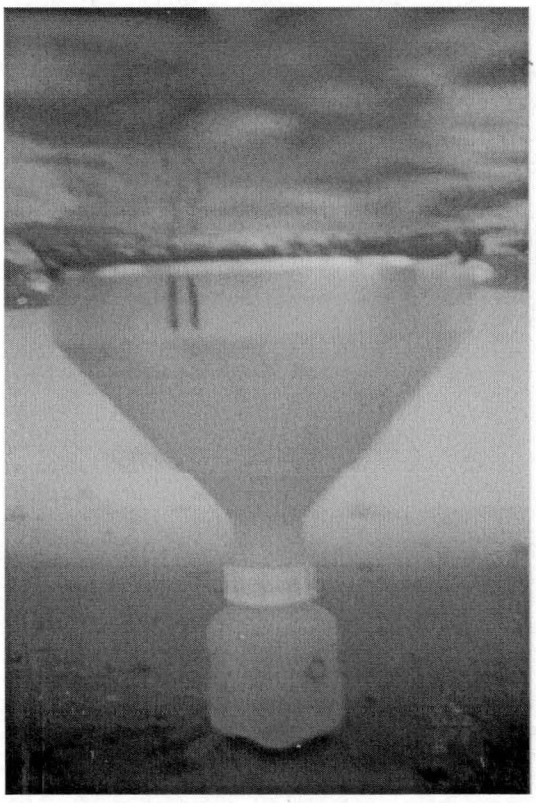

Fig. 4.3 Sediment trap set under an oyster basket. Traps secured under basket using elastic chord.

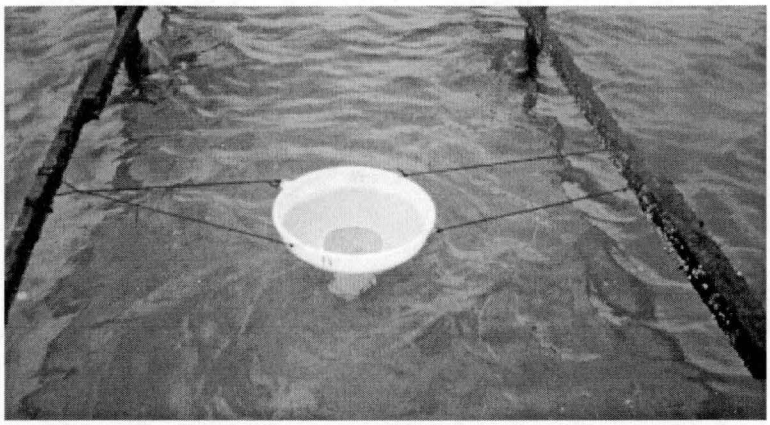

Fig. 4.4 Control trap set between oyster rack rails.

On return to the laboratory each day, the material in the jars was allowed to settle before carefully siphoning off the overlying water. The weight of total particulate matter (TPM) collected was determined following oven drying of the material at 65^0 C for 24-48 hours. The percentage of particulate organic matter (POM) was calculated by the difference in weight following ashing in a furnace at 480^0 C for 2 h. Daily values of biodeposits (faeces and pseudofaeces combined) were determined from the weight of total particulate matter collected in the traps under oysters, less the control trap values.

On the final day of each sampling period, 5 replicate sediment cores were collected from under each trap using a 60 ml syringe (core ID 26 mm) with the tip removed, and the samples combined. Generally the oxic layer, which was quite visible, was shallow and consequently only the top 0.8 to 1 cm was collected for particulate organic matter determinations by loss on ignition (480^0 C for 2 h) after oven drying at 65^0 C overnight. An assessment was made of combustion time and hence loss of organic matter for sand type sediments, which showed no significant difference ($P > 0.001$) between 2 h and 4 h combustion times, thus 2 h was considered to yield representative results.

4.2.2 Tidal volumes and flow study

During the June sampling (winter), water samples were collected from three sites along a transect across the lagoon near Bens Gutter (Fig. 4.1). Buoys were set to mark the location of each site, and water samples were collected from 1 m depth sub–surface in 2 L bottles at approximately 1½ h intervals during the outflowing and incoming tide during the day. Temperature, salinity and secchi depth was measured at the central site

only due to the narrow width of the channel. Particulate matter quality and quantity, chlorophyll a and nutrients (nitrate, phosphate and silicate) were measured in each replicate sample at each sampling time. However, only the particulate matter data has been used here.

A tide gauge was stationed near the north–east corner of the study lease (Fig. 4.1) and recorded tide height at 30 minute intervals. Details of the tide gauge and method of calculating volumes and flow are described in Chapter 3. Tidal velocity was calculated from the total volume of water flowing in or out of the body of the lagoon beyond the transect for each half hour period. The mean surface area of the lagoon beyond the transect was 4.605×10^6 m^2 and the mean cross–sectional area of the transect was 517.8 m^2.

The seston mass at each stage of the tide sampled was calculated by multiplying the concentration of total particulate matter by the calculated volume of water flowing in or out across the transect. The duration of the tidal study each day was 6 hours (commencing 9.30 am 15/6/96 and 10.20 am 16/6/95).

4.2.3 Statistical analysis

One–way analysis of variance (ANOVA) was conducted using Genstat™ Version 3.2 to assess differences in total particulate matter (TPM) and percentage particulate organic matter (%POM) between trap types (under oysters or controls) and in sediments between the two sampling periods (summer 1995 and winter 1996). The ANOVA tables of the statistical comparisons are presented in Appendix 2.

4.3 Results

4.3.1 Biodeposition

At each of the sampling periods the lease was being used to on–grow and condition oysters of 60 – 70 mm size. Total length of racking for the lease was 9.5 km. For oysters of this size, the general density is 60 oysters per basket (unit) with 6 units per m of racking (Fig. 4.2). This equates to a density of 360 oysters m^{-2}. Estimates of the number of oysters held at each time were calculated (Table 4.1).

Table 4.1 Summary table of water temperature, biodeposition rates, background sedimentation rates and oyster numbers. (dw = dry weight).

	Summer '95	**Winter '96**
Water temperature (C)	17.0^0	9.3^0
Mean TPM (oysters–controls) g dw basket^{-1} day^{-1}	17.36 ± 7.41	3.81 ± 1.27
Mean TPM controls g dw day^{-1}	0.70 ± 0.39	0.77 ± 0.42
Mean daily biodepostion g dw m^{-2} day^{-1}	180.46	39.60
Mean background sedimentation g dw m^{-2} day^{-1}	7.28	8.00
Lease occupancy	50 %	85 %
No. of oysters held (10^6)	1.71	2.9
Mean daily deposition from lease (kg dw day^{-1})	494.8	184.6
Average daily biodeposition/oyster (g dw day^{-1} oyster^{-1})	0.29	0.064

Water temperature was 17^0 C in summer and 9.3^0 C in winter. During summer, 50% of the lease was stocked with most baskets located within the northern region of the lease, where they were closer to the inflowing water and hence better water quality in terms of incoming available food. During winter, 85% of the lease was stocked with oysters distributed over much of the lease area.

The area of the traps was 0.0962 m^2, thus the average daily biodeposition rates based on the mean TPM (g basket^{-1} day^{-1}) expressed as grams m^{-2} were, 180.5 g m^{-2} day^{-1} in the summer period and 39.6 g m^{-2} day^{-1} in winter. The control trap values for these same times were 7.28 g m^{-2} day^{-1} and 8.00 g m^{-2} day^{-1} respectively. Average biodeposition rate per oyster also varied at each time with a much higher rate measured in summer (0.29 g dw oyster^{-1}) than in winter (0.064 g dw oyster^{-1}). The total mean daily biodeposition for the study lease for the two time periods (Table 4.1) were quite different, with 494.8 kg day^{-1} deposited in summer and 184.6 kg day^{-1} in winter.

The mean TPM, or biodeposits, collected in the traps under oysters (less control trap values) and the control traps for the two periods (Fig. 4.5 a and b) showed considerable differences between oysters and control trap values for each period. Greater variation in daily biodeposition rates were shown over the three sampling days in summer than in winter (Fig. 4.5). Strong winds occurred at the time of the summer sampling than in winter. However, the values for the control traps at each of these times are similar, despite the water column instability.

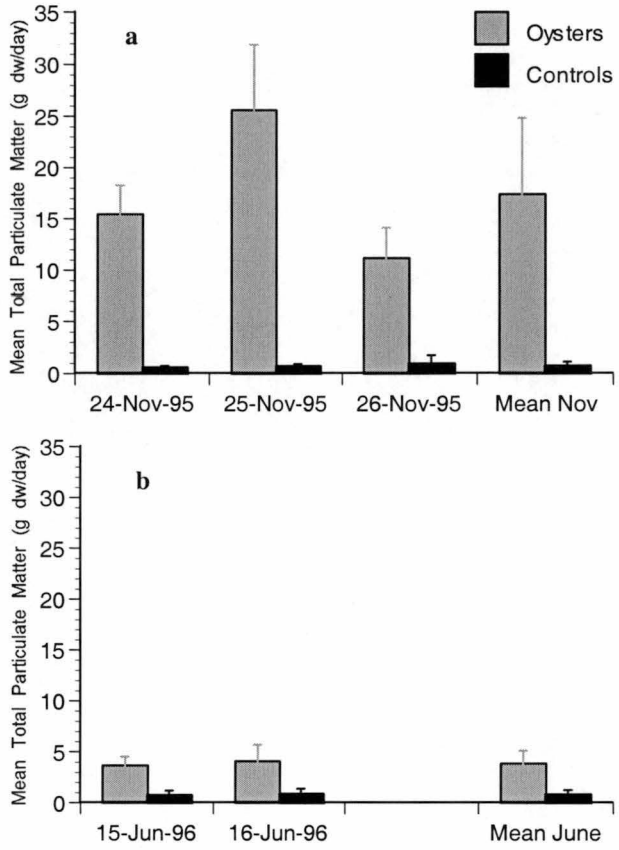

Fig. 4.5 Mean (+ sd) daily total particulate matter (g dw day^{-1}) of biodeposits collected from oyster traps (less control trap values) and material collected in control traps for the time periods a) summer and b) winter.

The amount of TPM collected in the traps showed significant differences between oysters and controls traps in summer ($P<0.005$) and winter ($P=0.001$). The quantity of material collected in summer was also significantly higher than that collected in winter ($P<0.001$). No significant differences were shown between the daily TPM values collected in traps for each period (Appendix 2).

The percentage particulate organic matter (%POM) of the material collected in the traps and in the sediment samples collected under each trap showed considerable differences (Fig. 4.6). Some variation was shown in the %POM of material collected in the control traps on each day in summer and winter. However, no significant difference was shown between the oyster and control traps in summer, though the differences between the two trap types was highly significantly different in winter ($P=0.005$). The mean %POM content of material collected under oyster baskets in summer and winter were similar. The %POM content of the sediments under oyster baskets and at control sites was low (< 2.6%), with the exception of one site sampled in winter (%POM = 12.25).

Comparison of the %POM content of sediments under oyster baskets and at control sites showed no significant difference at each sampling period (Appendix 2).

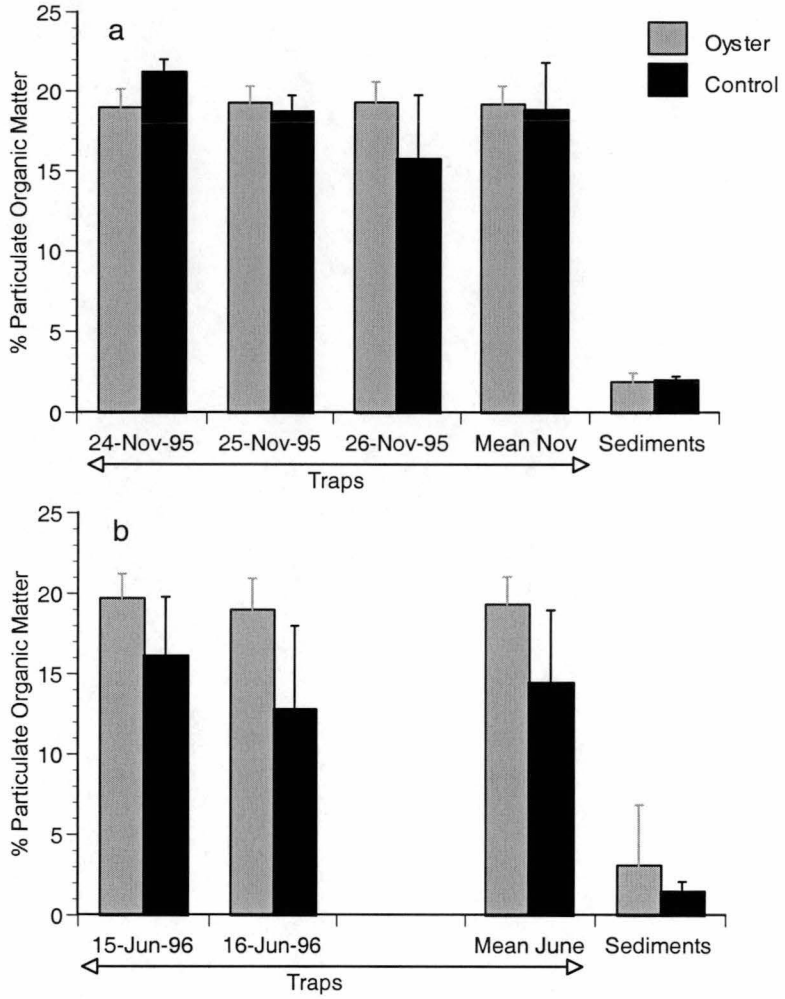

Fig. 4.6 Mean (+ sd) percentage particulate organic matter (%POM) of biodeposits, material collected in control traps and sediment samples under oyster and control traps for the time periods a) summer and b) winter.

4.3.2 Tidal velocity and flux

A tide gauge was positioned near the lease for approximately 6 months. Only the June 1996 data was analysed for this study. Results of the tide data for the three day period of the winter sediment trap study have been plotted (Fig. 4.7) and show the tidal range relative to the mean tide height for the cross–sectional area of the transect sampled. Tidal range over the three days was approximately ± 60 cm.

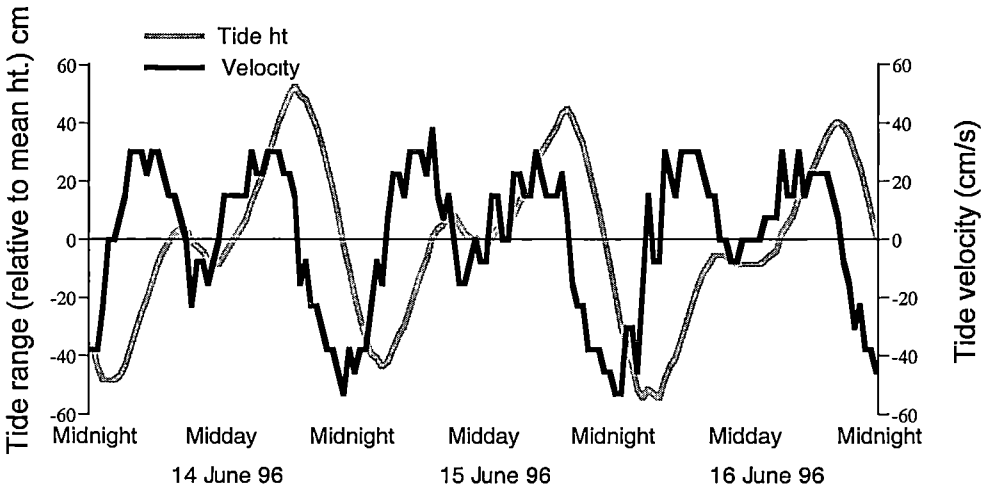

Fig. 4.7 Tide range (cm) and tidal flow velocity (cm s^{-1}) for three day period June 1996.

Tidal velocity was calculated for the total volume of water flowing in or out of the body of the lagoon beyond the transect for each half hour period. Whilst the velocity was variable over the half hour time intervals during the ebb and flood tides, generally faster velocities occurred during the ebb (outflowing) tides. This was also the general trend in the data over the June period. Mean tidal velocity across the transect for the three day period was 20.6 cm s^{-1} and for the whole of June was approximately 24 cm s^{-1}.

From the water samples collected across the transect, the mass of total particulate matter (seston) was calculated by multiplying the mean concentration of TPM (Appendix 2, Table 2.10) by the volume of water flowing in or out across the transect. The tide height for the 24 h period of each sampling day is shown (Fig. 4.8). The tidal range during the day was quite small. The calculated mass of seston ranged between 22 and 35 tonnes, with lower levels recorded at the time of low water.

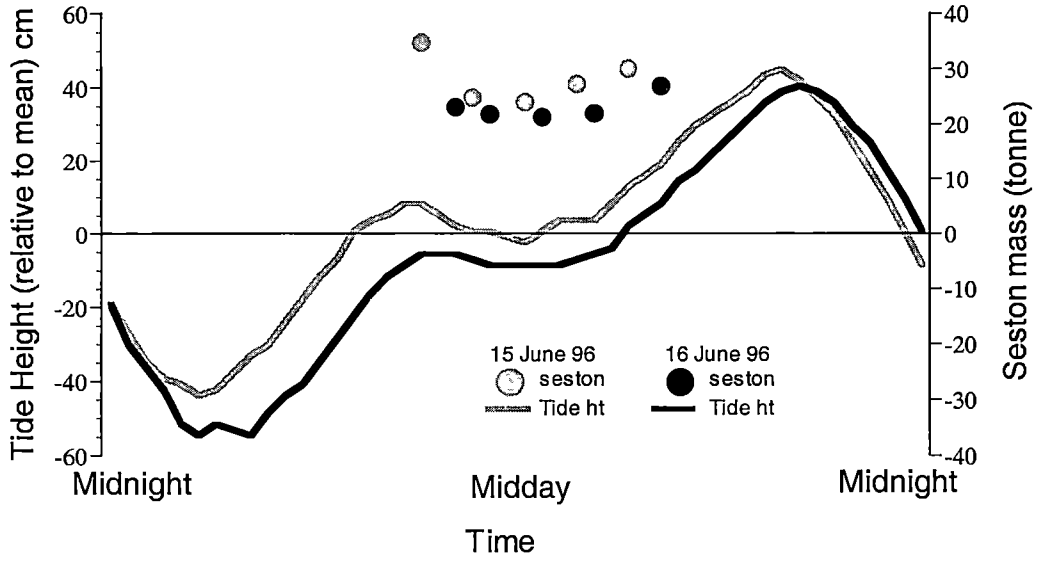

Fig. 4.8 Tide height and seston (TPM) mass (tonnes) across the lagoon transect over a two day period June 1996.

4.4 Discussion

Differences in biodeposition rates between the two periods were shown from material collected under baskets containing the same density of oysters of similar size. Biodeposition rates during the summer sampling period were 4 to 5 times greater than the winter values. However, this can not be simply attributed to temperature differences, as a number of factors affect biodeposition rates. Filtration rates and activity have been shown to vary with temperature, with generally reduced filtering activity at low temperatures (Walne, 1972; Winter, 1978; Powell et al., 1992) and hence biodeposition. Haven and Morales–Alamo (1966) noted an 85% reduction in oyster biodeposition rates when water temperatures decreased to 6.7 C, with further reductions below this temperature. The water temperature measured during the winter period of this study at Pipeclay Lagoon was approximately half that measured during the summer period (9.3^0 and 17.0^0 respectively). Whilst water temperature may have influenced biodeposition rates, it is most likely that other factors in combination were responsible for the differences.

The stocking rates of the lease differed in each of the sampling periods, with greater numbers of oysters held during the winter period (Table 4.1). During the summer period, lease occupancy was approximately 50% with most baskets located within the northern region of the lease to provide greater exposure to the incoming water and hence

increased food availability. Farm management experience has shown this spatial arrangement to induce greater growth rates (pers. comm. Rex Richardson, oyster farm manager). Logically it follows that greater numbers of oysters within a fixed area would result in reduced feed availability (depletion), and hence reduced intake per oyster due to greater competition of available food resources from the water column as it passes across successive racks on the lease. Studies have shown a reduction in available food, as measured by chlorophyll a or particulate matter concentration, across natural shellfish beds or culture areas (Asmus and Asmus, 1991; Navarro et al., 1991; Dame et al., 1992; Prins et al., 1994; Boyd and Heasman, 1998; Heasman et al., 1998).

Sornin et al. (1983) found a high correlation between the amount of seston in the water column and biodeposition rate, with seasonal variation in biodeposition related to the amount of particulate matter in the water. It is difficult to correlate biodeposition rates with seston quantity or quality, unless ejected and rejected material are collected separately. Pseudofaeces are produced as a consequence of rejection of particles (selective ingestion) or when particulate matter concentration exceed a threshold value (Kiørboe and Møhlenberg, 1981; Shumway et al., 1985; Powell et al., 1992; Barillé and Prou, 1993; Pastoureaud et al., 1996; Hawkins et al., 1997).

Many studies have reported the uptake (assimilation, clearance) of food, frequently measured by chlorophyll a or particulate matter, and compared this to ambient levels (e.g. Navarro et al., 1991; Hawkins et al., 1996). Measurements and comparison of biodeposits (faeces and pseudofaeces, either combined or separate) with available food, has shown variable results from enrichment, marginal alteration, or reduction in quality. Studies conducted by Hawkins et al. (1996; 1997) showed variable results from organic enrichment of ingested material with inorganic matter rejected, to rejection of matter similar to the seston composition beyond a threshold concentration at which the mussels (*Mytilus edulis*) were able to utilise. In the Bassin de Marennes–Oléron (France), Pastoureaud et al. (1996) showed high levels of degraded chlorophyll a products (phaeophorbides) in the pseudofaeces of oysters (*Crassostrea gigas*), indicating selective rejection of degraded matter in the seston, which possibly was biodeposits resuspended in the water column.

Average %POM in the sediment traps under the oysters was similar between the two time periods (summer and winter), though the average %POM in the control traps was similar to the oyster traps in summer but significantly different in winter. Water

samples collected near the study lease in Pipeclay Lagoon at approximately the same time as the experimental periods, showed variation in chlorophyll a, total particulate matter and %POM values. Average chlorophyll a, TPM and %POM measured during the tidal flux study in winter were 0.76 $\mu g\ L^{-1}$, 7.42 $mg\ L^{-1}$ and 31.34% and in summer were 1.94 $\mu g\ L^{-1}$, 26.87 $mg\ L^{-1}$ and 26.20% respectively, as measured in Bens Gutter (site 3). These results are from water samples collected around the low tide from sites close to the study lease. Brown and McCausland (1999) reported similar average chlorophyll a concentrations from a site near the mouth of the lagoon, collected at a fixed time each sampling day during similar times to this study. Calculated %POM from these authors results, however, were generally higher (38.22–50.83%) than those measured during this study (26.20 - 31.34 %).

If it is assumed that the water column %POM values measured at these times are representative of the available food to the oysters, then it appears that there is a reduction in the organic fraction of the particulate matter by the oysters (in the course of filtration and selective ingestion). Using the seston %POM figures above and the mean %POM content of biodeposits, the estimated mean absorption efficiency calculated by the Conover method as adopted by Crawford et al. (1996), was 33.3% in summer and 47.6% in winter. However, Iglesias et al. (1998) caution that unless faeces and pseudofaeces are collected separately, such calculations overestimate the amount of food ingested and absorbed. They suggest that when biodeposits (faeces and pseudofaeces) are collected together, the efficiency estimates represent food processing efficiency rather than absorption. In this study, the degree of pseudofaeces production was unknown as the traps collected all material deposited from under each basket with no separation.

Interestingly, the control traps showed similar %POM values as the oyster traps in summer, and suggest that material collected in the traps most likely was resuspended biodeposits. During the winter sampling, the %POM of material collected in the control traps was significantly different to the oyster traps and quite different to the background levels measured in the water samples. This suggests that material collected in these traps was possibly resuspended sediments, or refiltered biodeposits. Kautsky and Evans (1987) noted similar observations of reduced organic content between biodeposits and naturally sedimented material during rough weather conditions.

Variation in TPM collected in the traps under oysters was shown over the three day sampling period in summer (November 1995), but little variation shown over the two day period in winter (June 1996). Strong winds occurred during the summer sampling, with calmer conditions during the winter sampling. However, little variation was shown in the %POM of the oyster traps over the sampling days at each time with greater variation shown in the control traps. This may have been linked to the water column instability and hence resuspension of sediments and most likely biodeposits on occasions. Haven and Morales–Alamo (1966) reported variation in weekly and daily biodeposition rates, with fluctuations in daily rates exceeding changes measured weekly and monthly. These authors also found the organic matter content in seston was approximately 40% higher than that measured in biodeposits. The estimates calculated for Pipeclay Lagoon were approximately 27% and 38% higher in summer and winter, respectively. Kautsky and Evans (1987), however found the mean organic content of biodeposits to be higher than the naturally sedimented material (30.4 ± 2.2% and 24.8 ± 1.5% respectively).

A consideration of this study was, how do these results compare with those obtained elsewhere? Few studies have been conducted by which direct comparisons can be made and these have been summarised in Table 4.2 below, with calculations made on occasions from available data to obtain tabulated estimates.

Table 4.2 Biodeposition rates of shellfish.

Location	Species	Density m^{-2}	Natural Sed g dw $m^{-2}d^{-1}$	Biodeposition g dw $m^{-2}d^{-1}$	Biodepn/oys or mus day^{-1}	TPM mg L^{-1}	Source
Canada	M e, M t	400	36.4	88.7	0.22 g	5 – 20	Hatcher et al. (1994)
Northern Baltic	M e		1 – 37	0.52–9 76	1 – 16 mg		Kautsky and Evans (1987)
British Columbia	C g			5 7	0 12 – 0 22 g	19.8 – 47.5	Bernard (1974)
Marennes–Oléron (F)	C g	200		480 – 6000	2.4 – 30 g		Sornin et al (1983)
York River, USA	C v	62		14 3 – 34 6	0.23 – 0.56 g		Haven and Morales–Alamo (1966)
Newfoundland	M m				4 3 – 40.9 mg		Navarro and Thompson (1997)
Bay of Morlaix (F)	C g			0.066 – 0.246			Boucher and Boucher–Rodoni (1988
Tasmania	C g	360	7.3 – 8 0	39 6 – 180.5	0.064 – 0 29 g	7 4 – 26 9	Mitchell (this study)

Natural sed = natural sedimentation (or control) values, oyst = oyster, mus = mussel, SPM = suspended particulate matter, M e = *Mytilus edulis*, M t = *Mytilus trossulus*, C g = *Crassostrea gigas*, C v = *Crassostrea virginica*, M m = *Modiolus modiolus*

The data in Table 4.2 show variation in the degree of natural sedimentation and biodeposition rates expressed as either g dw m^{-2} day^{-1} or per animal. Results from *Crassostrea gigas* in this study at Pipeclay Lagoon are comparable to those obtained by Bernard (1974) in British Columbia and for *C. virginica* in the York River (Haven and Morales–Alamo, 1966) for oysters of similar size. The variation in biodeposition results expressed as g dw m^{-2} day^{-1} is possibly because of differences in the density of animals per m^2. Interestingly, the rates reported for mussel species were much lower, with the exception of those from Canada (Hatcher et al., 1994). These authors reported similar TPM and daily biodeposition levels for mussels, though the natural sedimentation rate was approximately 5 times higher than that measured in Pipeclay Lagoon. The values reported for *C. gigas* in the Bassin de Marennes–Oléron (France) by Sornin et al. (1983) are high and most likely due to a number of factors. A number of studies related to shellfish culture have been conducted within the Bassin de Marennes–Oléron, where the tidal amplitude is large (approximately 5 m), there are extensive areas of oyster culture (Raillard and Ménesguen, 1994) and seston loads have been reported to be high (50 – 200 mg L^{-1}) (e.g. Grant and Bacher, 1998). In light of the high TPM loadings reported from Marennes–Oléron Bay, it is most likely that pseudofaeces form a greater component of the oyster biodeposits.

Many of the studies cited in Table 4.2 reported seasonal variation in biodeposition rates, principally related to temperature and seston, or phytoplankton abundance (Haven and Morales–Alamo, 1966; Sornin et al., 1983; Kautsky and Evans, 1987; Hatcher et al., 1994), with generally reduced rates in winter and elevated rates in the warmer months. Similar observations were made with this study in Pipeclay Lagoon, though more extensive sampling would be needed to confirm this trend.

The results of the sediment samples show low organic matter levels (mean range 1.46 – 3.07 %POM), quite different to the levels measured in the sediment traps both under oyster baskets and at control sites, with the exception of one site where the organic matter content was elevated (12.25%). Sediments in this area of the lease (north–east corner) were observed to be more loosely compacted and suggestive of a depositional region. These results are similar to those measured by Thorn (1997) (1.14%), though sediment %POM content at a reference site outside of oyster culture areas was found to be lower (0.64%). This reference site, however, was located near the main channel for the body of the lagoon, and may well have been subjected to greater scouring effect.

Results of the sediment samples suggest that material is not accumulating under the racks, but is being transported and deposited elsewhere. Tidal velocities calculated were frequently of sufficient speed to cause transportation, and on occasions, erosion of the sediment boundary layer (Day, 1981d) and hence biodeposited material. Widdows et al. (1998) report a threshold current velocity required to resuspend biodeposits (faeces and pseudofaeces) of 15 – 20 cm s^{-1}. Kusuki (1978), in a study of transport rate of oyster faeces during settling from 0.5 m above the substrate, reported deposition to the substrate at 2 m distance with a current velocity of 1 cm s^{-1}, 5 m distance at 2 cm s^{-1}, and at velocities of 5.5 cm s^{-1} biodeposits were retained in suspension and transported elsewhere. Sinking rates of oyster faeces ranged from 0.4 to 1.8 cm s^{-1} (Kusuki, 1978). Tidal velocities, calculated from the June tide gauge data, were frequently greater than this speed and these results were considered to be reflective of annual current velocities experienced within the lagoon.

Studies elsewhere have shown organic enrichment under shellfish culture as a result of increased sedimentation (or depositional) rates, generally measured as elevated organic matter content or organic carbon (Dahlbäck and Gunnarsson, 1981; Grenz et al., 1991; Barranguet et al., 1994; Feuillet–Girard et al., 1994; Grant et al., 1995; Barranguet, 1997). Feuillet–Girard et al. (1994) reported a six fold increase in organic carbon under oyster (*Crassostrea gigas*) cultivation areas as compared to control sites. In a study of mussel culture in Nova Scotia conducted by Grant et al. (1995), sedimentation rate was found to be approximately 2.3 times higher under mussel culture than at reference sites. However, organic carbon content of the sediment was found to be similar. These authors attributed this to the fact that mussel biodeposits are derived from phytoplankton, and hence had a similar carbon content to the sedimenting material found at the reference site. Dahlback and Gunnarsson (1987) noted similar observations, with sedimentation rate 3 times greater under mussel culture compared to control sites, though the C/N ratio of the material collected was similar. Interestingly, these studies have shown that organic enrichment of the sediments is not reflected by organic carbon or nitrogen content, but rather by the degree of sedimentation.

Some studies have shown sediment redox potential measured under shellfish culture to be reduced (Sornin et al., 1990), with anoxic sediments observed (black colouration), elevated sulphur/sulphide levels (Dahlbäck and Gunnarsson, 1981; Sornin et al., 1983), and at times mats of the bacteria *Begiattoa* (Dahlbäck and Gunnarsson, 1981). No redox measurements were recorded on the sediments within the Pipeclay Lagoon study

lease, however, visual observation of the sediment samples showed the oxic layer to be shallow (less than 1 cm), with grey sediments and faint hydrogen sulphide odour detectable beyond this depth. Similar sediment profiles have been observed elsewhere in the lagoon both within (pers. obs.) and near shellfish culture areas (Mitchell and Macleod, 1998). *Begiattoa* was not observed within the study lease, nor has it been observed (in my experience over 10+ years) at any intertidal shellfish farm within Tasmania. This could be due to the shallow nature of intertidal culture areas in Tasmania, in which the hydrodynamics are predominantly wind driven, or tidally driven, or a combination of these two main influences. Hence sediment resuspension plays an influential role on sediment dynamics, and is most likely responsible for the reduced opportunity for *Begiattoa* mats to develop.

Management controls have been stipulated for intertidal shellfish culture by the Department of Primary Industry, Water and Energy (DPIWE – formerly DPIF) which state that "there must be not more than 1 km of stocked racking per hectare of lease area" (DPIF, 1998). Thus, on this basis and from the information on lease areas within Pipeclay Lagoon (DPIF, 1998), approximate figures of biodeposition rates were calculated for the lagoon, based on the stocking rate of the study lease and estimation of maximal production from the other leases. Total lease areas (less the study lease) is 35.39 ha, and the estimated area of racking of the other leases was calculated to be 35.39 km. The daily load of biodeposits from all the leases during the summer period, calculated from the g m^{-2} figures (Table 4.1), was estimated to be 7.24 tonnes dw day^{-1} and 1.72 tonnes dw day^{-1} in winter. The background sedimentation load for the lagoon (using the average area of 460.5 ha) was calculated to be 33.49 tonnes day^{-1} in summer and 36.8 tonnes day^{-1} in winter (using control trap values). Whilst these figures are crude estimates, the results of the tidal flux study indicate that some material (biodeposits and natural seston) is being retained within the lagoon. This suggests that this material is either being deposited or utilised by filter feeders within the lagoon. It should be noted that the tidal flux study was conducted during a tide with little variation in amplitude. It is likely that greater rates of material would be transported or exported from this region of the lagoon during tides with greater amplitude. However, more extensive sampling over differing tides (e.g. spring and neap tides) would be needed to assess this.

Additionally, there is a large wild population of feral Pacific oysters and mussels in some regions of the lagoon. In recent times, regular removal of feral Pacific oysters is

undertaken by the marine farmers as a community service (due to community objections to feral Pacific oysters), but also to reduce competition by these filter feeders on available food resources within the lagoon. Recently, approximately 10–20 tonnes of feral Pacific oysters were removed from the lagoon, particularly from the region north of the main lease areas. Since the start of this program in 1991, an estimated 500 tonnes have been removed (source: The Mercury, 23 April 1999).

Concern has been raised with respect to the impact of shellfish biodeposition on macroalgae, in particular seagrass species and potential decrease of these beds. In Pipeclay Lagoon seagrass, predominantly *Heterozostera tasmanica,* occurs in the deeper channel towards the mouth of the lagoon, but isolated beds occur elsewhere in the lagoon. In recent times, these beds have been noted to have increased (pers. obs.). Survey of regions adjacent to lease areas within Pipeclay Lagoon (Mitchell and Macleod, 1998) showed a general substrate type of fine sand with variable amounts of shell debris, however regions with reduced current flow (predominantly within the deeper channel areas) showed greater amounts of silt with a more loosely compacted substrate surface. Observations within a seagrass bed (*Heterozostera tasmanica*) located nearby to shellfish culture structures, showed entrapment of sediment having a consistency of black fine sand/silt with hydrogen sulphide odour. It is possible that biodeposits from the shellfish lease areas are being transported and deposited in the channel regions or deep hole located at the south end of the lagoon. Entrapment of this material by the seagrass beds could have a beneficial role in providing nutrients to the rhizosphere, as has been shown by Reusch et al. (1994).

5. Oyster growth and condition

5.1 Introduction

The culture of Pacific oysters (*Crassostrea gigas*) in Tasmania has rapidly increased since the establishment and supply of hatchery reared spat in the late 1970s. The majority of oyster farms in Tasmania are intertidal, where oysters are grown in mesh baskets, or envelopes, on fixed wooden racks. Growth and condition of oysters can, to a large extent, be regulated by the height that they are held on the rack structures and hence immersion period. Generally, the industry standard is approximately 40% exposure time, but this can vary within the range of approximately 10-50%.

Growth rate and condition is principally governed by temperature, food quality and quantity, hydrodynamic influences (e.g. transport/supply of food), stock density, and management practices (e.g. stocking rates, frequency of grading, farm design). The presence, location and size of other farms within an area can also significantly influence growth rates and condition by competing for available food resources. This has been shown in mussel long line cultures (e.g. Navarro et al., 1991; Heasman et al., 1998) and in studies across natural oyster and mussel reefs (e.g. Asmus and Asmus, 1993; Smaal and Zurburg, 1997).

Bivalve growth rates and condition indicies have frequently been used as a means of assessing conditions within an area with respect to environmental factors, site suitability and carrying capacities (e.g. Hickman et al., 1991; Almeida et al., 1997; Almeida et al., 1999; Toro et al., 1999). Many of these studies have shown correlation of temperature, salinity, chlorophyll a and particulate matter with growth rate and condition of oysters (Brown and Hartwick, 1988a, 1988b; Cigarría, 1999; Almeida et al., 1999; Toro et al., 1999) and mussels (Rodhouse et al., 1984; Page and Hubbard, 1987; Hickman et al., 1991; Stirling and Okumus, 1995).

A complicating aspect which needs to be factored into these assessments is, differences due to somatic growth and reproductive growth (gametogenesis). That is, diversion of energy and hence changes in condition due to reproductive development and subsequent loss following spawning. Oysters are not sold at this time because meats are poor in quality and appearance and are of unacceptable market quality. After spawning, animals

convert food ingested to glycogen which is stored (Quayle, 1969). This is reflected in an increase in soft body tissue weight post spawning, and generally occurs over the autumn to winter period.

However, if food supply is low, or of poor quality, or prevailing environmental conditions unfavourable, oysters will draw on their glycogen reserves. Unless this is replenished, body weight will decline and this is reflected in lower condition indices. During the colder winter period, metabolic activity and filtration rates are reduced (Walne, 1972; Winter, 1978; Powell et al., 1992), with little change in growth shown. As water temperature increases with the onset of spring, filtration rates and hence feeding activity increase and glycogen content, and hence body tissue weight, increase relative to available food supplies (Quayle, 1969). Under favourable conditions, reproductive tissue replaces glycogen in preparation for spawning. However, the amount of reproductive tissue developed is influenced by the amount of glycogen stored preceding this time (Quayle, 1969). This process is closely linked to temperature, with spawning in *Crassostrea gigas* reported to occur in water temperatures ranging from 18.5^0 - 24^0 C (Medcolf and Wolf, 1975).

The soft body of oysters are covered by the mantle, which is responsible for the formation and growth of shell (Quayle and Newkirk, 1989). The mantle generally remains in contact with the inner surface of the shell. However, if the soft body mass declines for prolonged periods, the mantle compensates for the increased inner space by laying down shell. This increases the shell thickness and shell weight relative to the whole weight of the animal. The shell of oysters are composed predominantly of calcium carbonate (~95%) (Shaw, 1969). The deposition of calcium carbonate is influenced by the concentration in the water and duration of immersion (Dame, 1996). The concentration of calcium carbonate in water is strongly influenced by salinity, with much greater levels found at higher (marine sea water) salinities (Day, 1981b). These factors influence shell growth and weight in oysters.

Trends in oyster growth rates and condition within areas can be assessed from prevailing environmental characteristics of those areas, such as food quality and quantity and hydrodynamic regimes. Difference in type and rates of growth with respect to dimensional aspects (shell length, width and depth), live weight, shell weight and meat weights are influenced by the environmental characteristics of an area. They also reflect

the suitability of an area for shellfish culture, the degree of competition experienced and hence indication of the effect of overall stocking rate, or influence of farms nearby.

Frequently shell length and live weight are measured, however, shell width and depth additionally provide valuable information with respect to environmental factors which influence growth and shape (often referred to as the 'cuppiness' of an oyster). The condition of shellfish is measured by various condition index methods, as a means of assessing the health and quality of animals. This is of particular relevance to the market quality of farmed product. There is considerable variability in condition indices used and much debate as to recommendation of an appropriate method(s). Lucas and Beninger (1985) reviewed and conducted evaluations of the most commonly used condition indices in bivalve studies. These authors recommended a simple, easily measured method, and one which would be readily standardised, for assessing the physiological state of an animal. This was based on the ratio of dry tissue weight to dry shell weight. Low values would indicate poor environmental conditions, or loss of condition due to spawning (Lucas and Beninger, 1985).

Similarly, Crosby and Gale (1990) conducted a review and evaluation of various condition indices using volumetric and gravimetric measures. They concluded that condition index methods based on volumetric displacement to measure shell cavity capacity, could introduce large sources of error due to the difficulties in accurately measuring displacement volumes - citing that reporting displacement volumes to ± 1 ml translates to errors of ± 1 g dry meat weights. These authors recommended gravimetric measurement of shell cavity capacity as the difference in whole live weight and dry shell weight.

Pacific oyster (*Crassostrea gigas*) growth and condition was assessed from studies conducted over three consecutive periods from March 1995 to December 1996 on commercial oyster farms in Pitt Water, Pipeclay Lagoon and Little Swanport. Relationships between oyster growth and condition, and the water quality and environmental parameters measured for much of this time period, will be discussed in the final chapter.

5.2 Materials and methods

Oyster growth trials were conducted at two sites on one farm in Pitt Water, Pipeclay Lagoon and Little Swanport (Fig. 5.1, Fig. 5.2 and Fig. 5.3), representative of average growth conditions (as selected by the oyster farmers).

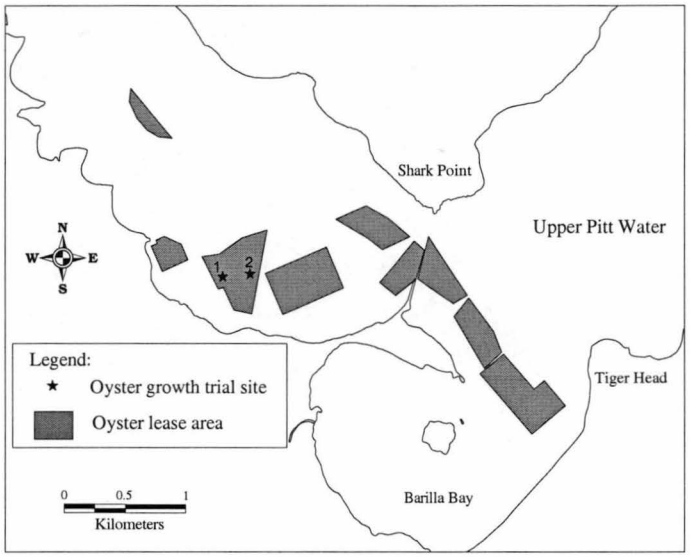

Fig. 5.1 Approximate location of oyster growth trial sites in Pitt Water. Site 1 = Inside, Site 2 = Farside.

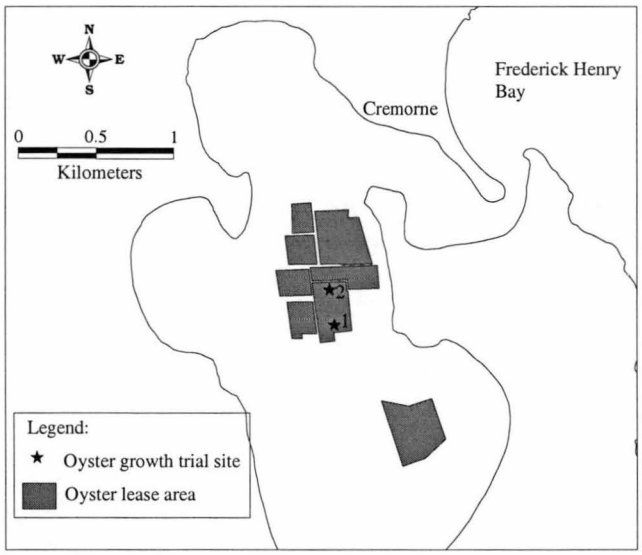

Fig. 5.2 Approximate location of oyster growth trial sites in Pipeclay Lagoon. Site 1 = South side, Site 2 = North side.

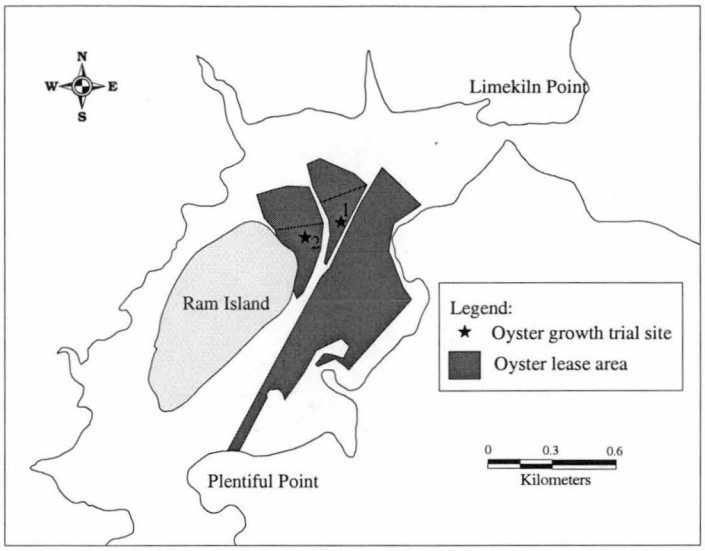

Fig. 5.3 Approximate location of oyster growth trial sites in Little Swanport. Site 1 = Channel, Site 2 = Ram Island.

Oysters used in the trials were standard graded single oysters of approximately 50-60 mm size (shell length) provided by the oyster farmers, and were grown in plastic mesh baskets similar to those used on the farm. At each site, oysters were placed in 4 baskets at a density of 65 oysters/basket, the standard density used by industry for oysters of that size. At the beginning and end of each growth period (trial), 30 oysters per basket (120 per site) were randomly selected and measured for shell length, width and depth (Fig. 5.4) to the nearest mm using Vernier calipers, and whole (live) weight to the nearest 0.1 g.

Prior to measuring, the oysters were washed to remove sediment and debris, and epibionts were removed with a knife. Each basket was labelled with a plastic tag to enable comparison of growth within as well as between sites in each area. On occasion, oysters were transported to the Marine Research Laboratory and held in a re-circulating filtered seawater system overnight prior to measuring. The number of mortalities were also recorded.

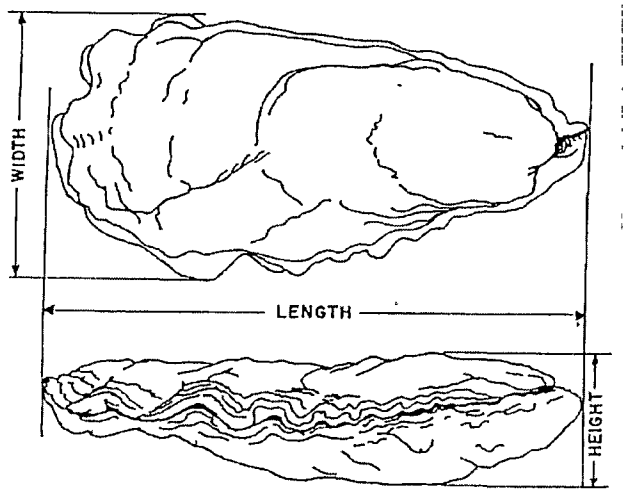

Fig. 5.4: Representation of the oyster shell dimensions measured for length, width and height (referred to as depth in this study) (Quayle and Newkirk, 1989).

Oyster condition was measured at the beginning and end of each growth trial. Initial condition was measured on a randomly selected sample of 40 oysters from the oysters provided by the farmer at the start of each trial. Final condition was measured using 10 oysters (selected at random) from each basket at each site (40 oysters/site). Condition indices (CI) were calculated using the following two methods:

Crosby and Gale Index (1990) = $\frac{\text{tissue dry weight (g) x 1000}}{\text{internal shell cavity capacity (g)}}$

where, internal shell cavity capacity = whole live weight (g) – dry shell weight (g) (in air)

and

Lucas and Beninger Index (1985) = $\frac{\text{tissue dry weight (g)}}{\text{dry shell weight (g)}}$

Shell length, whole (live) weight, tissue dry weight (80^0 C for 48 h) and shucked shell dry weight (80^0 C for 24–48 h) were measured.

It was anticipated that the oyster growth trials would be conducted over 6 month periods, with new oysters used for each trial. However, it was not possible to maintain this consistent experimental schedule, though new oysters were used for each trial. The actual dates and time intervals (days) are given in Table 5.1. During the first trial at Pitt

Water, oysters were measured at approximately 3 months and again at approximately 6 months. Oysters removed for condition measurement at the 3 month period, were replaced with oysters from the same batch grown in a fifth replicate basket at each site. Thereafter, all trials were conducted on undisturbed oysters for the duration of the trial periods.

Table 5.1 Date and time interval (days) of oyster growth trials at Pitt Water, Pipeclay Lagoon and Little Swanport.

Area	Lease No.	Trial No.	Start Date	End Date	No. days
Pitt Water	48	1	21 March 95	29 June 95	100
		1	29 June 95	17 October 95	210
	48	2	17 October 95	22 March 96	157
	48	3	22 March 96	3 October 96	195
Pipeclay Lagoon	15	1	16 March 95	19 September 95	187
	15	2	20 September 95	8 March 96	169
	15	3	8 March 96	27 September 96	203
Little Swanport	86	1	30 March 95	27 November 95	242
	86	2	27 November 95	22 May 96	177
	86	3	22 May 96	17 December 96	209

5.2.1 Data analysis

Assessment was made of trends in the growth of oysters, using length, width and depth measurements, for each trial period. Mean percentage increases were determined by comparison of the mean initial and final measurements.

The relationship between shell length to live weight was assessed, using data from the initial and final measurements. All measurements recorded from the four baskets at each site were combined for this analysis, and plotted to obtain the power curve equation of the form $L = aW^b$, where L is the shell length (mm), W is whole (live) weight (g), a and b are constants (King, 1995).

A nested two–way Model 1 analysis of variance (ANOVA) was used to assess variation between baskets within sites (nested, fixed factor) and between sites (fixed factor) for each trial period. The design of the ANOVA table for each parameter of growth (length, width, depth and weight) and condition (length, weight, shell cavity volume, dry meat weight, dry shell weight, Crosby and Gale CI, Lucas and Beninger CI) is shown in Table 5.2.

Table 5.2 Design of the ANOVA table used to assess growth parameters (length, width, depth and weight) and condition (weight, shell cavity volume, dry meat weight, dry shell weight, Crosby and Gale CI, Lucas and Beninger CI).

Source (for Growth)	Degrees of freedom	F
Site	a - 1 = 1	$F_{(1,6)} = MS_{site}/MS_{basket(site)}$
Basket(Site)	a(b-1) = 6	$F_{(6,232)} = MS_{basket(site)}/MS_{error}$
Error (oysters)	ab(n-1) = 232	
Total	239	

where a = 2 sites, b = 4 baskets/site, n = 30 oysters/basket

Source (for Condition)	Degrees of freedom	F
Site	a - 1 = 1	$F_{(1,6)} = MS_{site}/MS_{basket(site)}$
Basket(Site)	a(b-1) = 6	$F_{(6,72)} = MS_{basket(site)}/MS_{error}$
Error (oysters)	ab(n-1) = 72	
Total	79	

where a = 2 sites, b = 4 baskets/site, n = 10 oysters/basket

Calculated F values were compared to expected F values (Zar, 1999) and level of significance determined (ie $\alpha = 0.05$, 0.01 or <0.001).

Plots of the data from the condition index measurements are provided using the overall site mean values. Mean instantaneous daily length, live weight and dry meat weight growth at each area over the three trial periods were calculated. Mean instantaneous daily growth (k) was calculated using

$$k = \frac{Ln_2 - Ln_1}{t} \quad \text{(Gerdes, 1983)}$$

where Ln_2 = mean final length, live weight or dry meat weight, Ln_1 = mean initial length, live weight or dry meat weight and t = duration of the trial period (days).

5.3 Results

5.3.1 Oyster growth

5.3.1.1 Pitt Water

The growth trials were conducted over the approximate seasonal periods of winter 1995 (trial 1), summer 1995/96 (trial 2) and winter 1996 (trial 3). Marginal change was shown in the growth of oysters at one site (site 2) throughout the 210 day period of the first trial (trial 1), with the exception of total weight and depth (Table 5.3). The sites, which had been selected by the oyster farmer, were not representative of average growth, but rather extremes of growth for the lease area. An alternative site, more

representative of average conditions was selected within the same region of the lease for the subsequent trials.

Greatest increase in growth was shown by weight changes for each period, which ranged from approximately 31% to 105%. Weight change was greatest at both sites during the second trial period, which was conducted over the summer period. Increase in shell length was low in comparison to the increases in width and depth.

Some variation was noted between the sites by the state of the oysters, or baskets, on retrieval. At the end of trial 1, the oysters from site 2 were cleaner with less mud/silt covering than oysters from site 1. At the end of trial 2, the baskets at site 2 had a reasonable amount of barnacle growth, but little present on the baskets at site 1.

Table 5.3 Mean initial and final length, weight, width and depth of oysters from two sites in Pitt Water (all baskets at each site combined). (n = 120/site) NB: The 3 month results of Trial 1 not included.

			Site 1 (Inside)				Site 2 (Farside)			
Trial	Time		Length (mm)	Weight (g)	Width (mm)	Depth (mm)	Length (mm)	Weight (g)	Width (mm)	Depth (mm)
1	Initial	Mean	64	26.1	35	22	65	26.9	36	22
		sd	*5*	*5.4*	*3*	*3*	*6*	*4 8*	*3*	*3*
	Final	Mean	72	40.4	43	25	64	35 3	35	24
		sd	*7*	*7*	*5*	*3*	*5*	*6.5*	*3*	*3*
		% increase	13	55	21	14	-1	31	-2	6
2	Initial	Mean	69	25	38	22	69	26 8	39	22
		sd	*4*	*4 2*	*4*	*2*	*4*	*4*	*4*	*2*
	Final	Mean	81	51 4	46	27	76	49.3	44	27
		sd	*7*	*8.3*	*4*	*3*	*6*	*7 5*	*4*	*3*
		% increase	17	105	22	27	11	84	14	22
3	Initial	Mean	74	38.7	41	26	74	38.8	41	27
		sd	*6*	*6.9*	*5*	*3*	*7*	*7 5*	*4*	*3*
	Final	Mean	89	58 2	54	29	92	61	58	29
		sd	*7*	*9.1*	*6*	*3*	*9*	*11 5*	*7*	*3*
		% increase	20	50	33	11	26	57	42	7

The relationship between shell length and live weight was similar at site 2 for the first and second trial periods (Table 5.4), and indicated greater shell length to live weight for smaller oysters and greater weight to shell length for larger oysters. However, the correlation ($R^2 = 0.14$) shown for site 2 during the first trial period indicates a poor relationship between shell length and live weight. During the final trial period (trial 3), the relationship between shell length and live weight were similar and indicated greater live weights for smaller oysters, and better correlation ($R^2 = 0.79$).

Table 5.4 Shell length (L mm) to live (whole) weight (W g) relationship of oysters from two sites in Pitt Water for each trial period.

	Site 1 (Inside)	R^2	Site 2 (Farside)	R^2
Trial 1	$L = 26.74\ W^{0.2691}$	0.55	$L = 39.88\ W^{0.1398}$	0.14
Trial 2	$L = 35.02\ W^{0.2119}$	0.64	$L = 39.06\ W^{0.173}$	0.47
Trial 3	$L = 17.65\ W^{0.3966}$	0.73	$L = 14.58\ W^{0.4468}$	0.79

Statistical comparison was made between sites at the start and finish of each trial (Table 5.5). This was to done to check that sizes of oysters used at each site at the start of each trial were not statistically different. Variation between baskets at each site, particularly at the end of each trial was assessed. However, comparison could not be made of differences between baskets at the final sampling of trial 2. At time of final measurement of the oysters, it was discovered that the oysters had been inadvertently emptied from the baskets by the farm employees prior to collection, mixed and then returned to the baskets. Whilst assurance was given that oysters from each site had been kept separate, the return of oysters to their respective baskets was highly unlikely.

No significant difference was shown in each of the parameters measured at the start of each trial, with the exception of trial 2, in which highly significant differences were shown between the width and weight of oysters at each site (P=0.01 and P=0.05, respectively). Significant differences were shown for each parameter measured between sites at the completion of trial 1, for reasons previously described. No significant difference was shown between baskets with the exception of weight, in which some variation (P=0.01) between baskets was evident (Table 5.5).

In trial 2, highly significant differences were shown in shell length between sites (P=<0.001), site 1 having the longer lengths. Shell widths were also significantly different (P=0.002). As previously described, differences were noted in the amount of barnacle fouling between these sites at time of collection. Considerable fouling was shown on the baskets from site 2 as compared to baskets from site 1. At the completion of trial 3, no significant differences were shown between sites for shell length, depth and weight, with significant difference evident in width (P=0.01) between sites. Baskets were relatively uniform with the exception of variation in length with some difference evident (P=0.05).

Table 5.5 Summary table of nested ANOVA results for shell length, width, depth and live weight of initial and final data at two sites, over the three trial periods at Pitt Water. NB. Only results of a one-way ANOVA are shown for final results of trial 2 (refer to text).

			Initial							Final						
Trial	Variable	Site	Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)	Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)
1	Length	1	64 08	ns	64 73	63 87	64 50	63 23	ns	72 43 a	<0 001	73 93	73 80	71 40	70 60	ns
		2	64 92		63 20	64 63	66 57	65 30		64 02 b		63 97	64 67	62 70	64 77	
	Width	1	35 35	ns	35 23	36 03	35 43	34 70	ns	42 6 a	<0 001	43 50	43 53	41 60	41 77	ns
		2	35 63		35 67	35 67	35 57	35 60		34 97 b		34 17	35 27	34 80	35 63	
	Depth	1	22 27	ns	22 13	22 73	22 17	22 03	ns	25 29 a	0 050	25 60	25 20	25 13	25 23	ns
		2	22 46		22 20	22 43	22 20	23 00		23 7 b		23 23	24 50	22 63	24 43	
	Weight	1	26 08	ns	26 36	26 36	26 33	25 27	ns	40 41 a	0 050	41 67a	42 36a	38 37b	39 25a	0 010
		2	26 89		24 69	26 87	28 09	27 90		35 34 b		32 86c	38 01a	33 9bc	36 59ab	
2	Length	1	69 20	ns	70 87	68 67	68 43	68 83	ns	80 78 a	<0 001					
		2	69 18		70 80	68 23	68 83	68 87		76 48 b						
	Width	1	37 95 b	0 010	37 93	37 63	38 03	38 20	ns	46 22a	0 002					
		2	38 93 a		39 20	39 10	38 47	38 97		44 38b						
	Depth	1	21 51	ns	22 30	21 80	20 70	21 23	ns	27 25	ns					
		2	22 08		22 47	22 17	21 90	21 80		26 96						
	Weight	1	25 03 b	0 050	26 02	25 24	24 11	24 74	ns	51 43	ns					
		2	26 79 a		27 96	26 73	26 16	26 34		49 27						
3	Length	1	74 25	ns	75 90	74 03	73 23	73 83	ns	89 20	ns	91 13a	89 4a	87 33a	88 93a	0 050
		2	73 52		73 27	75 83	72 80	72 17		92 33		93 47a	95 67a	91 77ab	88 4b	
	Width	1	40 80	ns	41 60	41 17	40 23	40 20	ns	54 46b	0 010	55 63	53 20	54 63	54 37	ns
		2	40 98		42 53	40 93	40 47	39 97		58 00a		59 03	58 03	58 37	56 57	
	Depth	1	26 02	ns	26 03	25 77	27 27	25 00	ns	28 79	ns	29 67	28 60	28 70	28 20	ns
		2	27 27		27 03	27 20	26 83	28 00		29 08		29 40	29 50	28 73	28 70	
	Weight	1	38 71	ns	39 64	38 17	39 21	37 81	ns	58 18	ns	59 37	59 05	57 72	56 58	ns
		2	38 77		39 73	40 35	37 33	37 67		61 01		62 60	63 65	60 45	57 33	

5.3.1.2 Pipeclay Lagoon

Similarly to Pitt Water, growth trials at Pipeclay Lagoon were conducted over the approximate seasonal periods of winter 1995 (trial 1), summer 1995/96 (trial 2) and winter 1996 (trial 3). The lease used for the trials was located along the western foreshore, approximately mid-way between two other operational marine farm leases. The two sites were effectively located on the upstream (site 2 - north) and downstream (site 1 - south) side of the farm with respect to incoming tidal flow.

Greatest percentage increase was shown in live weight at the end of each of the three trials, ranging from 42 -67% (Table 5.6). The oysters on the northern side (site 2) of the lease showed higher live weight increases during trial 1 and trial 3, corresponding to the winter period. Site 1 showed the higher live weight gains during the summer period of trial 2. The highest percentage increase was for width in each trial and at each site. On average, oysters at site 2 showed the slightly greater increase in dimensional growth (i.e. length, width and depth), and live weight, with the exception of trial 2. During this trial period, the greatest increase was live weight (42%), though percentage increase in shell length, width and depth were relatively small.

Table 5.6 Mean initial and final length, weight, width and depth of oysters from two sites in Pipeclay Lagoon (all baskets at each site combined). (n = 120/site)

			Site 1 (South)				Site 2 (North)			
Trial	Time		Length (mm)	Weight (g)	Width (mm)	Depth (mm)	Length (mm)	Weight (g)	Width (mm)	Depth (mm)
1	Initial	Mean	68	27 6	37	23	69	27 9	37	23
		sd	*6*	*4 4*	*3*	*3*	*6*	*4.7*	*3*	*3*
	Final	Mean	80	40.5	48	26	80	45 6	51	27
		sd	*8*	*7.5*	*8*	*3*	*8*	*8.7*	*6*	*4*
		% increase	19	47	30	14	16	64	37	19
2	Initial	Mean	67	27 4	40	22	69	28 9	40	23
		sd	*8*	*8 1*	*7*	*3*	*8*	*8.6*	*6*	*4*
	Final	Mean	80	45 8	49	26	72	40.9	43	24
		sd	*9*	*10.2*	*5*	*3*	*10*	*11.1*	*6*	*4*
		% increase	19	67	22	16	5	42	9	4
3	Initial	Mean	64	37 6	37	24	63	37.5	37	23
		sd	*5*	*5.2*	*3*	*3*	*6*	*6 3*	*4*	*4*
	Final	Mean	85	59 3	55	29	84	61.5	53	30
		sd	*7*	*8.4*	*6*	*3*	*8*	*10 6*	*6*	*4*
		% increase	32	58	47	23	34	64	45	28

The relationship of shell length (mm) to live weight (g) was similar at site 1 (trial 2) and site 2 (trial 1) (Table 5.7). Generally for all trials, the equations produced showed site 2 to have the marginally higher live weights for given shell lengths. Trial 3 oysters at both sites indicated smaller oysters for a given weight.

Table 5.7 Shell length (L mm) to live (whole) weight (W g) relationships of oysters from two sites in Pipeclay Lagoon for each trial period.

	Site 1 (South side)	R^2	Site 2 (North side)	R^2
Trial 1	$L = 17.856\ W^{0.405}$	0.63	$L = 23.199\ W^{0.3259}$	0.63
Trial 2	$L = 23.656\ W^{0.3174}$	0.66	$L = 29.673\ W^{0.2444}$	0.41
Trial 3	$L = 9.329\ W^{0.5365}$	0.77	$L = 10.854\ W^{0.4914}$	0.70

No significant differences were shown in each of the parameters measured between sites at the start of each trial (Table 5.8). Some variation was shown between baskets in lengths (P=0.05) for trial 1 and 3, depth (trial 1) and width (trial 3). However, these variations were not evident in the final sampling, with the exception of depth at the end of trial 1. Highly significant differences (P=<0.001) were shown in weight between sites at the end of trial 1, and length at the end of trial 2. Significant differences were also shown between sites in width (P=0.01) at the end of trial 2. At the end of trial 3, no significant differences were shown in each of the parameters measured between, or within, sites.

Table 5.8 Summary table of nested ANOVA results for shell length, width, depth and live weight of initial and final data at two sites, over the three trial periods at Pipeclay Lagoon.

			Initial							Final						
Trial	Variable	Site	Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)	Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)
1	Length	1	68 87	ns	70 1a	65 4b	70 87a	69 13a	0 050	80 21	ns	80 27	79 70	79 63	81 32	ns
		2	67 85		68 2a	68 97a	67 63a	66 6a		80 43		81 10	82 41	80 33	77 90	
	Width	1	36 88	ns	36 33	37 67	36 47	37 03	ns	50 55	ns	49 37	50 87	51 43	50 54	ns
		2	37 03		37 03	36 23	37 97	36 87		48 30		46 10	48 43	51 20	47 47	
	Depth	1	22 92	ns	22 47b	24 2a	22 8ab	22 2b	0 050	27 29	ns	27 4a	27 03ab	28 9a	25 71b	0 050
		2	22 67		22 8ab	21 8b	23 77a	22 33b		25 89		26 93a	24 5b	26 2a	25 93ab	
	Weight	1	27 86	ns	28 18	26 99	27 91	28 38	ns	45 62a	<0 001	47 36	44 39	46 25	44 41	ns
		2	27 61		28 57	27 25	27 46	27 15		40 45b		40 48	41 37	41 00	38 96	
2	Length	1	68 67	ns	67 13	69 27	69 33	68 97	ns	71 98b	<0 001	70 57	72 27	72 67	72 43	ns
		2	67 01		66 27	66 57	64 77	70 43		79 82a		81 73	78 47	77 93	81 13	
	Width	1	39 73	ns	38 17	39 93	41 10	39 70	ns	43 48b	0 010	42 07bc	44 83ab	41 07c	45 93a	0 050
		2	40 20		38 37	42 57	38 53	41 33		49 07a		50 17a	49 17a	48 3a	48 63a	
	Depth	1	22 83	ns	21 97	21 97	23 80	23 60	ns	23 67	ns	23 27	23 03	22 97	25 40	ns
		2	22 24		23 03	22 97	20 67	22 30		25 81		26 43	26 33	24 53	25 93	
	Weight	1	28 86	ns	28 28	29 97	28 28	28 91	ns	40 94b	0 050	41 51	41 19	41 33	39 74	ns
		2	27 42		26 16	28 16	23 89	31 48		45 82a		47 56	45 89	42 33	47 50	
3	Length	1	62 65	ns	64 40a	62 63ab	62 03ab	61 53b	0 050	83 32	ns	84 80	84 30	82 00	82 17	ns
		2	64 12		66 73a	64 43ab	62 07b	63 23b		84 77		84 87	86 60	84 10	83 50	
	Width	1	36 59	ns	37 13a	36 33ab	37 53a	35 37b	0 050	53 12	ns	53 93	50 97	53 83	53 73	ns
		2	37 28		38 43a	37 30a	37 17ab	36 23b		54 64		54 07	55 23	56 33	52 93	
	Depth	1	23 43	ns	24 10	23 97	23 07	22 60	ns	30 11	ns	30 10	30 93	30 80	28 60	ns
		2	23 83		25 00	23 03	23 93	23 33		29 37		28 83	29 07	30 10	29 50	
	Weight	1	37 51	ns	40 26	37 14	37 05	35 61	ns	61 53	ns	65 86	60 78	60 09	59 38	ns
		2	37 64		38 12	38 97	36 97	36 50		59 30		60 20	60 46	57 39	59 16	

5.3.1.3 Little Swanport

The growth trials at Little Swanport were of slightly longer duration, especially trial 1 (242 days), than those at Pitt Water and Pipeclay Lagoon. Trials were conducted over the approximate seasonal periods of autumn to late spring 1995 (trial 1), late spring to late autumn 1995/96 (trial 2) and late autumn to early summer 1996 (trial 3).

Percentage increases in growth were greatest at Little Swanport, compared to Pitt Water and Pipeclay Lagoon. Greater increases were shown in shell length, width and depth for each of the trials (Table 5.9), with similar sized percentage increases in growth in each of these dimensions shown. Weight gain ranged from 157-230 % and percentage increase in shell length were generally greater than 31%. It was also noted that for the initial oysters used at the start of each trial, the live weights were much smaller than those of the oysters used at Pitt Water and Pipeclay Lagoon. This suggested the oysters used in the trials at Little Swanport were most likely much younger.

At the end of trial 2, site 2 (Ram Island) oysters were observed to have greater frill growth than those from site 1. A reasonable amount of fouling (thick algal matting) was noted on the baskets and the oysters were covered in a layer of silt sediment at the end of trial 3.

Table 5.9 Mean initial and final shell length, live weight, width and depth of oysters from two sites in Little Swanport (all baskets at each site combined). (n = 120/site)

			Site 1 Channel				Site 2 Ram Is			
Trial	Time		Length (mm)	Weight (g)	Width (mm)	Depth (mm)	Length (mm)	Weight (g)	Width (mm)	Depth (mm)
1	Initial	Mean	59	21 1	38	20	59	20 6	38	20
		sd	*6*	*3 6*	*4*	*2*	*6*	*3.5*	*5*	*2*
	Final	Mean	84	60 4	56	29	91	68 2	62	30
		sd	*9*	*10 6*	*7*	*3*	*10*	*13.7*	*6*	*4*
		% increase	41	186	48	43	56	230	62	49
2	Initial	Mean	64	20.1	35	20	62	18.4	33	19
		sd	*5*	*3 9*	*4*	*3*	*6*	*3.7*	*4*	*2*
	Final	Mean	83	51 6	50	27	88	53 6	55	28
		sd	*8*	*8*	*6*	*3*	*9*	*9 8*	*6*	*3*
		% increase	31	157	43	37	42	192	64	46
3	Initial	Mean	53	16 5	35	18	53	17.6	37	19
		sd	*3*	*3.3*	*4*	*2*	*3*	*3 2*	*5*	*3*
	Final	Mean	76	44 8	54	27	76	45.8	54	27
		sd	*6*	*6.6*	*5*	*3*	*6*	*7 2*	*6*	*3*
		% increase	43	172	53	49	44	161	46	43

Good correlation was shown between shell length and live weight at each site for the three trial periods (Table 5.10), and indicate relatively uniform overall isometric growth of the animals. The relationship was similar at both sites for trials 1 and 3, with marginally greater shell length per live weight shown in trial 2.

Table 5.10: Shell length (L mm) to live (whole) weight (W g) relationships of oysters from two sites in Little Swanport for each trial period.

	Site 1 (Channel)	R^2	Site 2 (Ram Island)	R^2
Trial 1	$L = 21.221\ W^{0.3352}$	0.87	$L = 18.968\ W^{0.373}$	0.90
Trial 2	$L = 27.022\ W^{0.2856}$	0.79	$L = 24.079\ W^{0.326}$	0.85
Trial 3	$L = 21.217\ W^{0.3346}$	0.85	$L = 19.31\ W^{0.3551}$	0.87

No significant differences were shown between sites for each of the parameters measured at the start of each trial, with the exception of shell width (P=0.05) and live weight (P=0.05) for trial 2 (Table 5.11). Highly significant differences were shown in width (P=<0.001) and live weight (P=<0.001) between baskets in trial 3, and significant difference in width between baskets (P=0.05) in trial 1. However, these differences were not apparent at the final sampling.

Significant differences were shown between sites at the end of trial 1 in shell length (P=0.05), width (P=0.05) and live weight (P=0.05), with site 2 showing the greater increases. Only shell width was significantly different (P=0.05) between sites in trial 2, however, significant differences were shown between baskets. Shell lengths were highly significantly different between baskets in trial 1 (P=0.005) and trial 2 (P=<0.001), with widths (P=0.005) and live weights (P=0.01) significantly different in

trial 2. No significant differences were apparent between sites or baskets at the end of trial 3.

Table 5.11 Summary table of nested two-way ANOVA for shell length, width, depth and live weight of initial and final data at two sites, over three trial periods at Little Swanport.

		Initial						Final						
Variable	Site	Site	P (site)	Basket (Mean)				P	Site	P (site)	Basket (Mean)				P
		(Mean)		1	2	3	4	(basket/site)	(Mean)		1	2	3	4	(basket/site)
Length	1	59 23	ns	57 47	60 30	59 13	60 00	ns	83 54b	0 050	82 33a	82 6a	83 03a	86 2a	0 005
	2	58 74		56 87	57 40	59 87	60 83		91 37a		88 67bc	87 43c	92 77ab	96 6a	
Width	1	37 86	ns	38 2a	37 83a	37 3a	38 1a	0 050	56 12b	0 050	56 00	56 57	54 37	57 53	ns
	2	37 91		35 77c	37 30bc	38 37ab	40 2a		61 52a		61 50	60 10	62 43	62 03	
Depth	1	20 16	ns	19 77	19 50	20 93	20 43	ns	28 92	ns	29 13	28 93	28 40	29 20	ns
	2	20 19		19 40	20 67	20 37	20 33		30 11		29 23	30 47	31 23	29 50	
Weight	1	21 12	ns	20 67	21 82	20 95	21 06	ns	60 39b	0 050	60 16	59 34	57 87	64 18	ns
	2	20 65		19 67	20 21	21 20	21 51		68 22a		66 53	65 97	69 16	71 22	
Length	1	63 68	ns	64 13	63 67	62 67	64 27	ns	83 26	ns	82 77ab	82 5b	81 03b	86 73a	<0 001
	2	62 10		60 97	62 97	61 57	62 90		88 18		82 97b	86 63b	91 3a	91 83a	
Width	1	34 77a	0 050	35 20	34 80	34 03	35 03	ns	49 58b	0 050	49 8ab	47 97b	48 53b	52 03a	0 005
	2	33 48b		32 63	34 10	33 80	33 37		54 86a		52 03b	54 63ab	55 77a	57 00a	
Depth	1	19 73	ns	20 07	19 77	19 43	19 63	ns	27 02	ns	26 77	26 80	26 93	27 60	ns
	2	19 20		19 03	19 43	19 57	18 77		27 96		27 50	27 83	28 83	27 67	
Weight	1	20 10a	0 050	20 53	19 75	19 55	20 57	ns	51 58	ns	53 21ab	49 62b	49 35b	54 15a	0 010
	2	18 38b		17 46	19 59	18 75	17 74		53 62		49 85b	51 72b	56 50a	56 41a	
Length	1	53 44	ns	53 40	53 43	53 93	53 00	ns	76 48	ns	77 00	75 30	75 77	77 87	ns
	2	52 70		51 67	53 67	53 40	52 07		75 76		77 37	76 17	74 37	75 13	
Width	1	35 41	ns	37 10a	36 53a	33 70b	34 30b	<0 001	54 16	ns	55 07	53 03	53 43	55 10	ns
	2	37 03		37 77a	37 90a	38 10a	34 37b		54 10		54 67	53 63	53 00	55 10	
Depth	1	18 14	ns	18 90	18 33	17 50	17 83	ns	26 96	ns	27 43	27 13	27 13	26 13	ns
	2	19 16		19 63	19 73	18 90	18 37		27 32		28 17	27 20	27 40	26 50	
Weight	1	16 51	ns	18 13a	17 18a	15 31b	15 41b	<0 001	44 83	ns	45 86	45 06	44 69	43 70	ns
	2	17 58		17 92a	18 80a	18 26a	15 34b		45 84		46 38	44 60	47 30	45 06	

5.3.2 Oyster condition

5.3.2.1 Pitt Water

During trial 1, oysters were measured at 3 months then again at approximately 6 months. Assessment of changes over this time were made on the data collected for determining condition indices. The greater increase in shell length during trial 1 occurred over the first three months, corresponding to the autumn period (Fig. 5.5). Similarly, little change was shown in shell length at both sites during trials 2 and 3 (Fig. 5.5).

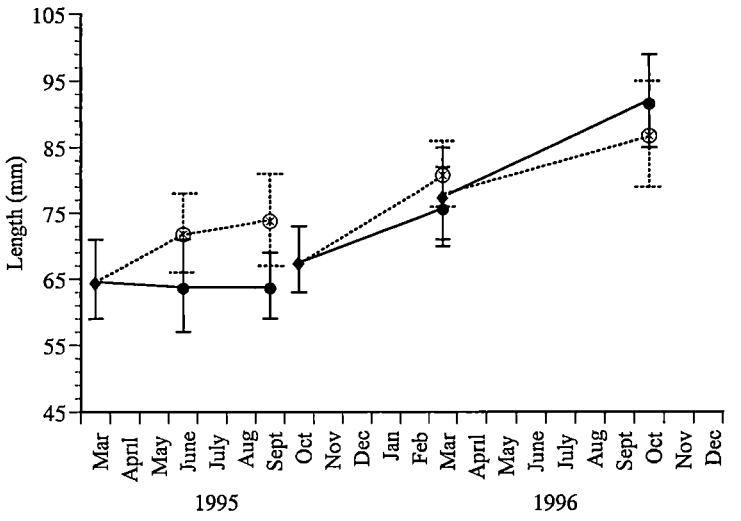

Fig. 5.5 Mean (± sd) shell length (mm) of oysters at two sites in Pitt Water over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

However, whilst oysters at site 2 showed no apparent change in length, live weight and dry meat weight increased (Fig. 5.6), particularly during the first three months. The percentage increase in dry shell weight and shell cavity capacity was greater over this same time (Appendix 3, Table 3.1). Growth between the three and six month period was considerably reduced at both sites, most likely as a consequence of coinciding with the colder temperatures. The greater increase in live weight and dry meat weight occurred over the summer period at both sites during trial 2 (Fig. 5.6). Little change was shown in dry meat weight at the end of trial 3, with greater increase in live weight, though the percentage increase was lower than that found in the previous trial (Fig. 5.6 and Appendix 3 Table 3.2).

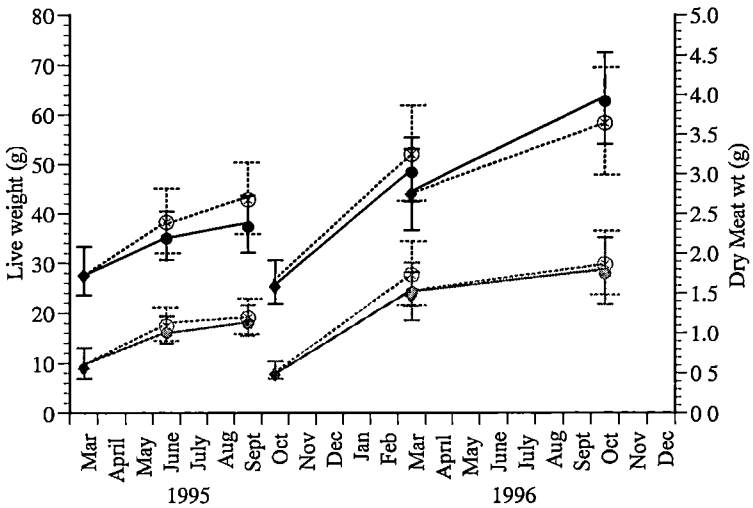

Fig. 5.6 Mean (± sd) live and dry meat weight of oysters at two sites in Pitt Water over three growth trials. Weight (g): ♦Initial, Final - ⊗ site 1, • site 2, Dry weight (g): ◊Initial, Final - ⊗ site 1, ◦ site 2 (n=40/site)

Condition indices of oysters, as measured using the Crosby and Gale and Lucas and Beninger condition indices were variable. Greater increase was shown in the values for Crosby and Gale condition index (CI) at the end of trials 1 and 2 at both sites, however lower increases were found in the third trial (Fig. 5.7) particularly at site 2. Interestingly, considerable variation was shown in the CI values at site 1 at three months. The Lucas and Beninger CI of oysters was lower at the end of the first trial, than at three months (Table 5.8). Greater increase was shown at the end of trial 2, which coincided with greater percentage increase in dry meat and live weight. The third trial showed a considerable reduction in Lucas and Beninger CI.

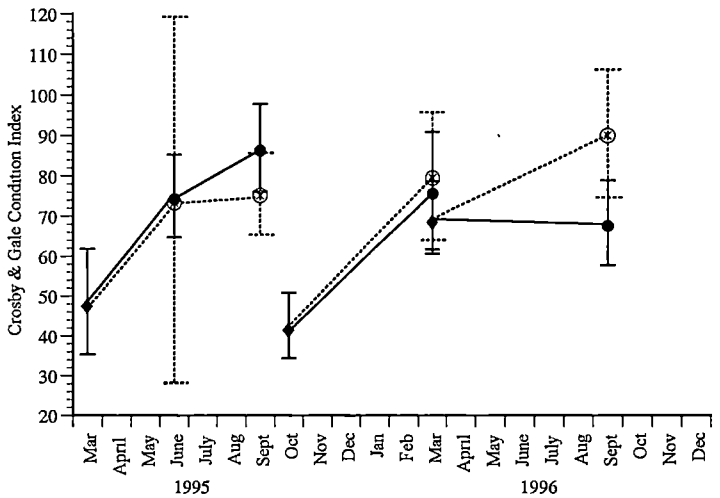

Fig. 5.7 Mean (± sd) Crosby and Gale condition index of oysters at two sites in Pitt Water over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

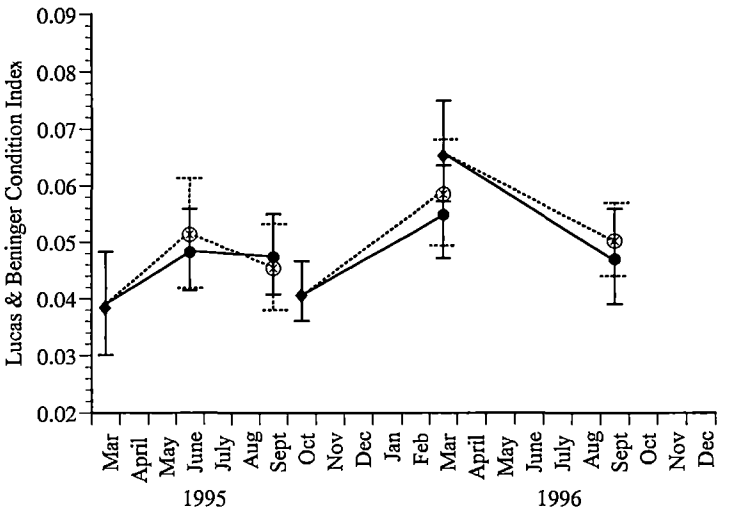

Fig. 5.8 Mean (± sd) Lucas and Beninger condition index of oysters at two sites in Pitt Water over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

Statistical analysis of the data showed differences between sites at the end of each trial. The summary table of these analyses are presented in Appendix 3 Table 3.4. In trial 1, significant differences were shown between shell length (P=0.005), live weight (P=0.05) and Crosby and Gale condition index (P=0.01). No significant differences were shown between baskets, with the exception of shell cavity capacity (P=0.05). In all parameters measured in trial 1, site 1 showed the greater change. Highly significant difference was shown in shell length at the end of trial 2 (P=<0.001), and moderately significant difference in dry meat weight (P=0.02), similarly site 1 showing the greater change. No significant differences were shown between baskets in the third trial, though significant differences were shown between sites. Shell length (P=0.05), shell cavity capacity (P=0.01) and Crosby and Gale condition index (P=<0.001) were significantly different between sites, with site 2 showing the greater change, with the exception of site 1 which showed the higher condition index.

5.3.2.2 Pipeclay Lagoon

Shell lengths at the end of trial 1 and 3 were similar between sites (Fig. 5.9). Greater variation was shown between the two sites at the end of trial 2. The percentage increases in shell length were smaller than those shown for live weight and dry shell weight in each of the trials (Appendix 3 Table 3.5). The increase in shell cavity capacity was similar for each site in trial 1 and 3, with a much higher increase shown only at site 1 in trial 2 (Appendix 3 Table 3.5). At the end of trial 2, site 2 showed smaller changes in each of the parameters measured.

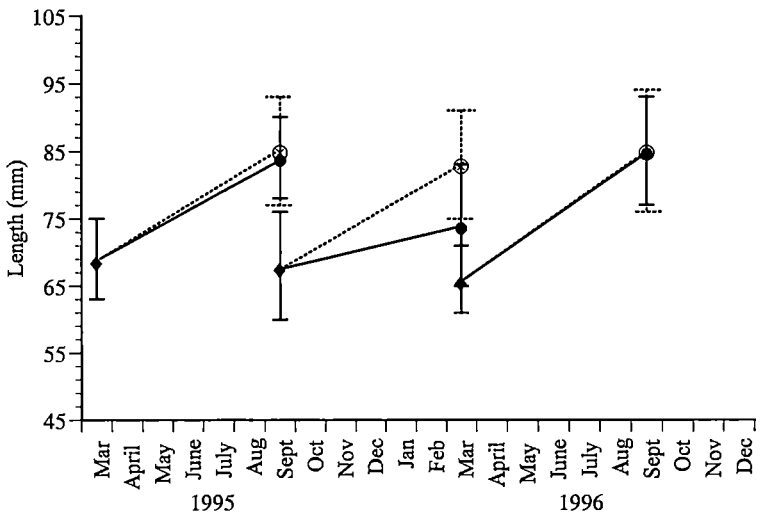

Fig. 5.9 Mean (± sd) shell length (mm) of oysters at two sites in Pipeclay Lagoon over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

Live weight increases were similar at the end of trials 1 and 3, with greater difference shown between sites at the end of trial 2 (Fig. 5.10). Dry meat weights increased over each of the trial periods, and showed a similar trend to the live weights. The percentage change in dry meat weight was similar to that shown for dry shell weight, except for site 1 in trial 3 (Appendix 3 Table 3.5). The percentage increase in dry meat weight at this site was approximately a third of that shown for live weight and dry shell weight.

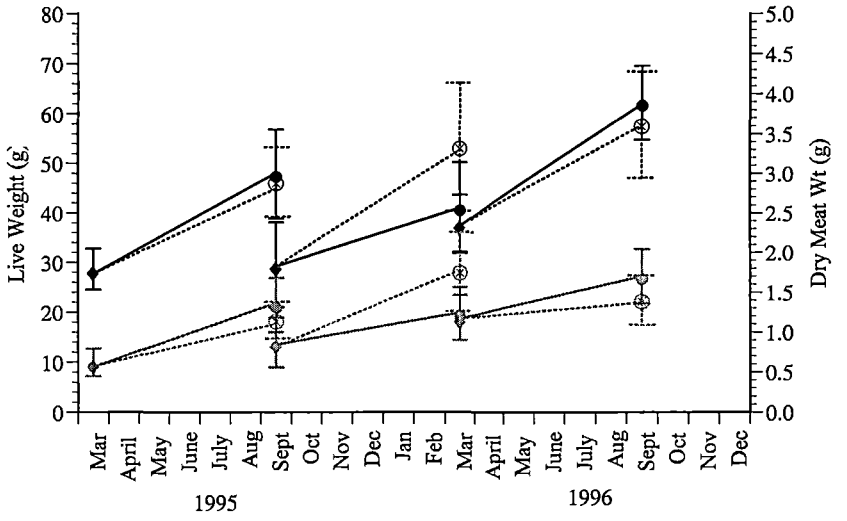

Fig. 5.10 Mean (± sd) live and dry meat weight of oysters at two sites in Pipeclay Lagoon over three growth trials. Weight (g): ♦Initial, Final - ⊗ site 1, • site 2, Dry weight (g): ♦Initial, Final - ⊗ site 1, • site 2 (n=40/site)

Oyster condition, as measured by the Crosby and Gale condition index (CI) increased over the duration of trials 1 and 2, but was less than the initial at the end of trial 3 (Fig. 5.11, Appendix 3 Table 3.6). Site 2 showed the greater CI values at the end of trial 1. A higher increase occurred in trial 2 at both sites.

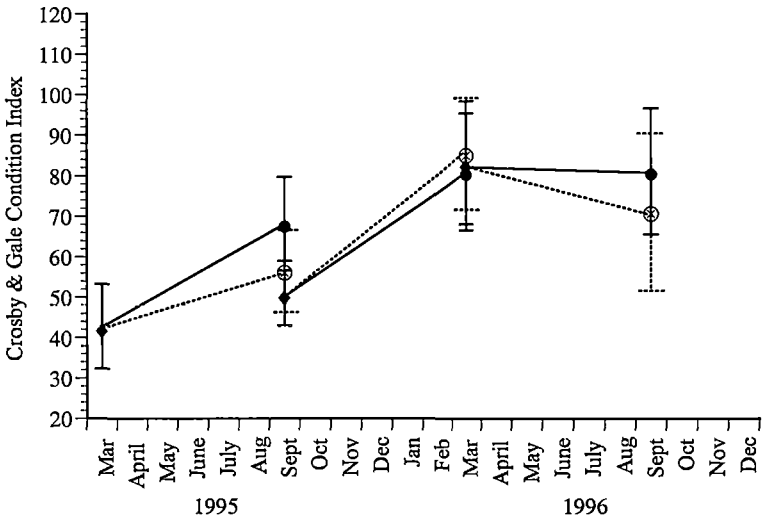

Fig. 5.11 Mean (± sd) Crosby and Gale condition index of oysters at two sites in Pitt Water over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

A similar trend was shown with the Lucas and Beninger CI, however the percentage change was considerably less than that found with the Crosby and Gale CI. Greater negative difference was shown at the end of trial 3, due most likely to the smaller relative change in dry meat weight.

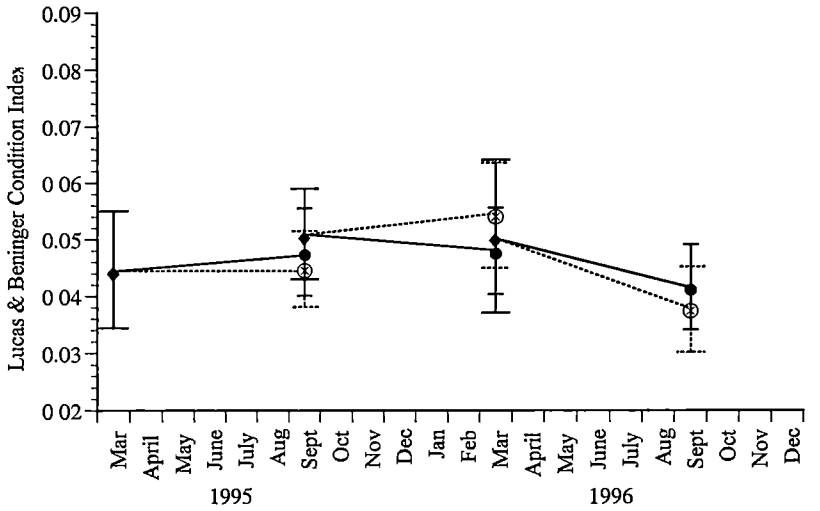

Fig. 5.12 Mean (± sd) Lucas and Beninger condition index of oysters at two sites in Pitt Water over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

No significant differences were shown between baskets at each site at the end of the three trials, with the exception of Crosby and Gale CI in trial 2 (P=0.05) (Appendix 3 Table 3.7). Dry meat weights were significantly different (P=0.05) between sites in trial 1 and 3, site 2 showing the larger weights. Oyster condition also varied between sites for these trial periods (P=0.05), but differed in the type of condition index measurement - Crosby and Gale CI in trial 1 and Lucas and Beninger CI in trial 3.

Highly significant differences were shown between sites at the end of trial 2 in shell length (P=<0.001), shell cavity capacity (P=<0.001) and live weight (P=0.01). Significant differences were also shown in dry shell weight (P=0.05), dry meat weight (P=0.05) and Lucas and Beninger CI (P=0.05). Site 1 showed the higher values.

5.3.2.3 Little Swanport

Considerable percentage increase were shown in shell length, live weight, dry shell weight, dry meat weight and shell cavity capacity (Appendix 3 Table 3. 8) at the end of each trial. These changes were markedly greater than those observed at Pitt Water or Pipeclay Lagoon. Increase in shell length was higher at site 2 (Ram Island) at the end of trials 1 and 2, with more comparable increase shown between the two sites at the end of trial 3 (Fig. 5.13).

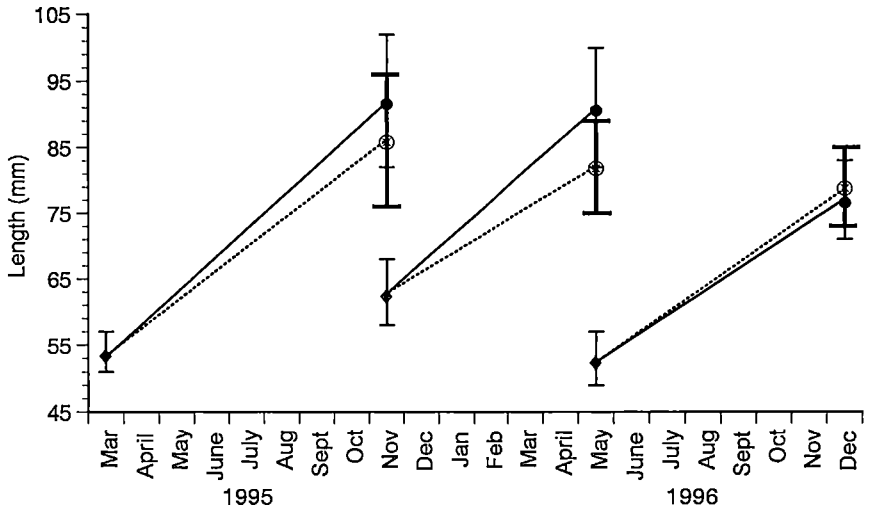

Fig. 5.13 Mean (± sd) shell length (mm) of oysters at two sites in Little Swanport over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

Increase in live weight and dry meat weight at the end of each trial, showed a similar trend between sites as found with the shell lengths. The magnitude of percentage increase in live and dry meat weight were similar at the end of each trial, though lower increase was shown over the summer period at the end of trial 2 (Fig. 5.14).

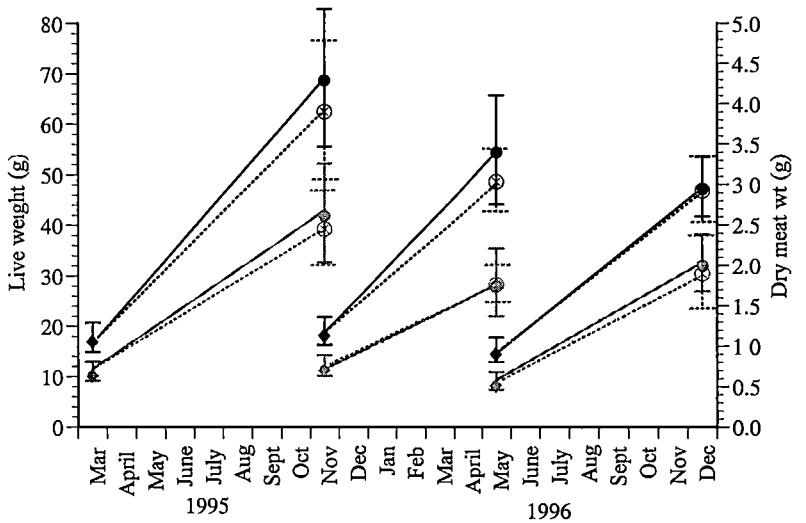

Fig. 5.14 Mean (± sd) live and dry meat weight of oysters at two sites in Little Swanport over three growth trials. Weight (g): ♦Initial, Final - ⊗ site 1, • site 2, Dry meat weight (g): ♦Initial, Final - ⊗ site 1, ◦ site 2 (n=40/site)

Oyster condition was reasonably high at the start of each trial (Fig. 5.15, Fig. 5.16, Appendix 3 Table 3.9) for both condition indices methods. At the end of trial 1 and 3, positive percentage increase were shown in Crosby and Gale CI, with a reduction shown at the end of trial 2, particularly at site 2 (Fig. 5.15).

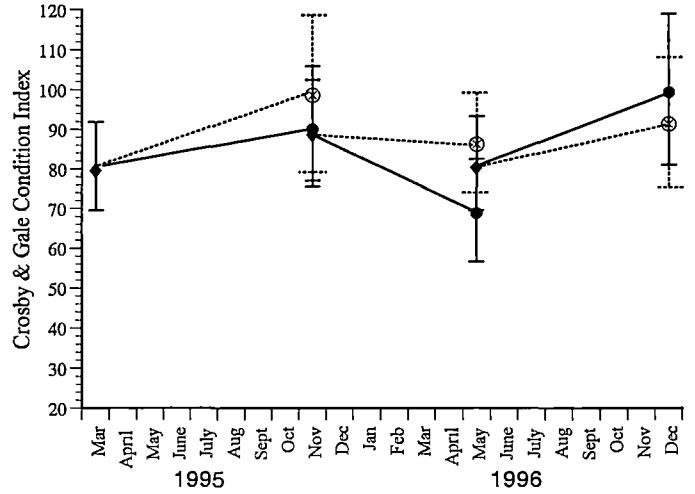

Fig. 5.15 Mean (± sd) Crosby and Gale condition index of oysters at two sites in Little Swanport over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

In contrast to the Crosby and Gale CI, a reduction was shown in the Lucas and Beninger CI at the end of trial 1 (Fig. 5.16). This most likely was due to the greater increase in dry shell weight, relative to dry meat weight. Similarly, a reduction was shown at the end of trial 2, with marginally greater increase shown at the end of trial 3. At the end of trial 3, a greater change was shown in dry meat weight than dry shell weight.

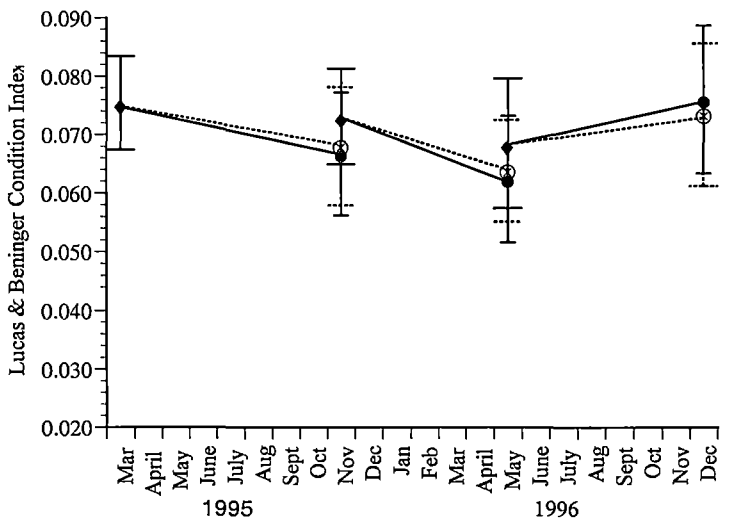

Fig. 5.16 Mean (± sd) Lucas and Beninger condition index of oysters at two sites in Little Swanport over three growth trials. ♦Initial, Final - ⊗ site 1, • site 2. (n=40/site)

No significant differences in the parameters measured were shown between sites at the end of trials 1 and 3 (Appendix 3 Table 3.10), with the exception of dry meat weight (P=0.05) in trial 1, site 2 having the higher values. At the end of trial 2, significant

differences were shown between sites in shell length (P=<0.001), live weight (P=0.05), shell cavity capacity (P=<0.001) and Crosby and Gale CI (P=0.05). Site 2 showed the higher values, with the exception of Crosby and Gale CI where oysters at site 1 had the higher index values.

Significant differences between baskets were apparent in the Crosby and Gale CI (P=0.01) in trial 2. Differences between mean values of the baskets at each site ranged from 78.5-91.9 at site 1, and 59.8-76.6 at site 2 (Appendix 3 Table 3.10). Highly significant differences were shown in this condition index value between baskets at the end of trial 3 (P=<0.001). Similarly, considerable variation in mean basket values were noted, ranging from 82.7-99.3 at site 1, and 82.7-115.6 at site 2. Dry meat weights between baskets were significantly different (P=0.05) at the end of trial 3, with variation shown (Appendix 3 Table 3.10).

5.3.3 Mean instantaneous daily growth

The mean instantaneous daily shell length, whole and dry meat weight growth rates varied between sites and trials (Table 5.12). Negative values were shown at site 2 at the end of trial 1 at Pitt Water. Values at the end of each trial at Pipeclay Lagoon were similar, with the exception of the lower mean daily shell length growths at site 2 in trial 2. Overall, the mean daily shell length growths were highest at Little Swanport.

Similarly, Little Swanport showed higher values for mean daily whole and dry meat weight growth at the end of each trial. However, the highest mean daily dry meat weight growth was shown at Pitt Water in trial 2. Mean daily dry meat weight growth was greater than whole weight growth in trials 1 and 2 at Pitt Water and Pipeclay Lagoon, with the reverse of this shown in both areas at the end of trial 3. Interestingly, despite the reduced shell length growth of oysters at site 2 (trial 1) at Pitt Water, mean daily dry meat weight growths were relatively high.

Table 5.12 Overall site mean instantaneous daily shell length (μm), whole and dry meat weight (mg) growth rates at two sites at the end of three trials in Pitt Water, Pipeclay Lagoon and Little Swanport.

Area	Trial	Site	Length (um)	Whole wt (mg)	Dry meat wt (mg)
Pitt Water	1	1	0 63	2.00	3.26
		2	-0.05	1.39	3.04
	2	1	1.10	4.39	7.44
		2	0.71	3.97	6.65
	3	1	0.54	1.38	1.11
		2	0.80	1.77	0.81
Pipeclay Lagoon	1	1	1.11	2.55	3.24
		2	1 02	2.73	4.09
	2	1	1.19	3.50	4.20
		2	0.47	1.96	2.07
	3	1	1.25	2 08	0.85
		2	1.23	2.45	1.80
Little Swanport	1	1	1.94	5.21	5.28
		2	2.19	5 62	5.57
	2	1	1.47	5.33	4.78
		2	2.08	5.98	4.81
	3	1	1.91	5.37	5 86
		2	1.78	5.43	6 13

5.3.4 Mortality

No mortalities were recorded at Pitt Water over the three trial periods. No mortalities occurred during the first trial at Little Swanport, with 1 mortality recorded from site 1 (Channel) in trial 2, and similarly in trial 3. At Pipeclay Lagoon, the only mortalities recorded occurred in trial 2, with 2 mortalities found in separate baskets at site 1 (South) and 1 mortality at site 2 (North). Overall, mortality was considered negligible, thus no analyses were conducted to assess mortality or survivorship relationships.

5.4 Discussion

Growth of oysters was greatest at Little Swanport with much greater percentage increases in shell length, live weight, width and depth. Growth at Pitt Water and Pipeclay Lagoon were similar. The sites within each farm were selected by the respective oyster farmers to be representative of average growth conditions. Significant differences were apparent between sites in each area at the final sampling, in shell length, width and weight. Generally, differences between baskets within sites were not significantly different, which is interesting considering the trials were conducted on undisturbed oysters. This suggests that grading of oysters at the size used in these growth trials may be performed at approximate six monthly intervals.

Farm management practice is to routinely grade oysters so that oysters of similar size are grown in the same basket. This process also breaks off the fine 'frill' growth to improve shell shape and condition (O'Meley et al., 1992). Oysters are graded using mechanical grading machines, which sort oysters by shaking on sieves of different mesh sizes. As oysters increase in size, the densities in baskets are reduced to ensure uniform growth, reduce competition for available food resources, and to minimise the effects of overcrowding which can lead to undesired shell shape (O'Meley et al., 1992). This process requires collection of the oyster baskets from the lease, bringing them in for grading and subsequent return to the lease, a process which requires additional handling and reduction in efficiency of farm operation.

The shape of oysters is important for market acceptability and consumer preference. The general accepted form is a relatively cupped shaped oyster, not too narrow in width relative to length. The shape of oysters used in this study was assessed at the start and end of each trial by determination of what has been referred to here as the aspect ratio, that is the width and depth relative to length. This was done to provide an easier means of assessing the uniformity of, or apparent trends in, the direction of shell growth overall. Whilst this comparison does not show differences in the overall size (i.e. shell length) of oysters, it does provide a means of determining the shape and changes which occurred over the duration of the growth periods (Table 5.13). In all cases shell width increased to varying degrees, while relative changes in depth were marginal to small by comparison. It appears that when greater ratio of width:length occurred, the ratio of depth:length was reduced. Oysters at Little Swanport showed proportional increases in width and depth ratios, suggesting more uniform (or isometric) growth. Nevertheless, the overall shape of oysters in each area were of acceptable market quality.

Table 5.13 Mean aspect ratio of shell length:width:depth of oysters at the start (initial) and end (final) of each of three trial periods at Pitt Water, Pipeclay Lagoon and Little Swanport (n=120)

Area	Trial	Initial Site 1	Final Site 1	Initial Site 2	Final Site 2
Pitt Water	1	1: 0.55: 0.34	1: 0.59: 0.35	1: 0.55: 0.34	1: 0.55: 0.38
	2	1: 0.55: 0.32	1: 0.57: 0.33	1: 0.56: 0.32	1: 0.58: 0.36
	3	1: 0.55: 0.35	1: 0.61: 0.32	1: 0.55: 0.36	1: 0.63: 0.32
Pipeclay Lagoon	1	1: 0.54: 0.34	1: 0.60: 0.32	1: 0.54: 0.33	1: 0.64: 0.34
	2	1: 0.60: 0.33	1: 0.61: 0.32	1: 0.58: 0.33	1: 0.60: 0.33
	3	1: 0.58: 0.38	1: 0.65: 0.34	1: 0.59: 0.36	1: 0.63: 0.36
Little Swanport	1	1: 0.64: 0.34	1: 0.68: 0.33	1: 0.64: 0.34	1: 0.67: 0.34
	2	1: 0.53: 0.31	1: 0.62: 0.32	1: 0.55: 0.31	1: 0.60: 0.32
	3	1: 0.70: 0.36	1: 0.71: 0.36	1: 0.66: 0.34	1: 0.71: 0.36

In each of the areas and trials, increase in live weight was far greater than increases in shell length, width or depth. However, an important factor is the degree by which the weight change was attributed to increase in meat weight, as opposed to shell weight caused by shell thickening (Rainer and Mann, 1992). The percentage of shell weight to whole weight was determined for each area at the start and end of each trial period (Table 5.14). Oysters at Pitt Water and Pipeclay Lagoon showed the higher values, indicating greater deposition of shell, though differences were apparent in the trial periods. At Pitt Water, the greater percentages of shell weight were shown at the end of trials 1 and 3, with lower values shown at the end of trial 2. Meat weights were much higher at the end of trial 2 and indicated favourable growth conditions. In trials 1 and 3, shell cavity capacity changes were small, and on occasion less than those initially measured. Greater change in shell length and width was shown at the end of trial 3, but small change in dry meat weights (17-24%).

The oysters at Pipeclay Lagoon showed a similar trend, with greater percentage shell weights at the end of trials 2 and 3. Greater percentage increase in live, shell and dry meat weights were shown relative to the increase in length at the end of trial 1, with much lower dry meat weights shown at the end of trial 3 for oysters of similar size. At Little Swanport, greater changes in shell, live, and dry meat weights were recorded, with similar order of magnitude shown in the percentage increases of each of these parameters. Increase in shell lengths were also much greater. The large dry meat weight and shell cavity capacity changes indicated favourable growth conditions with respect to soft body growth.

Table 5.14 Mean percentage shell weight to total live weight of oysters at the start (initial) and end (final) of three trials at Pitt Water, Pipeclay Lagoon and Little Swanport (n=40).

Area	Trial	Initial	Final Site 1	Final Site 2
Pitt Water	1	55.2	62.4	64.7
	2	50.1	57.1	57.6
	3	51.3	63.9	59.1
Pipeclay Lagoon	1	48.9	55.6	58.7
	2	57.5	61.3	62.8
	3	62.5	65.0	65.9
Little Swanport	1	51.5	58.5	57.3
	2	55.0	57.4	52.2
	3	54.4	55.4	56.6

A comparison of Pacific oyster growth at 10 locations in British Columbia, Canada, showed oysters at medium growth sites had significantly less dry meat weight in relation

to shell weight, as compared to high growth sites (Brown and Hartwick, 1988a). These authors attributed the differences to preferential partitioning of energy resources to increasing shell growth and thickness over soft body weight during prolonged conditions of low food supply. In comparison to soft tissue growth, shell growth and metabolic maintenance is more energetically efficient under conditions of prolonged low food supply (Brown and Hartwick, 1988b).

The Lucas and Beninger condition index provides an indication of the physiological status or health of bivalves at different times of the year, or growing conditions (Lucas and Beninger, 1985). These authors state that low values indicate that recent biological effort has been expended, either in reproductive growth, spawning (release of gametes), or maintenance under poor environmental conditions. The lower Lucas and Beninger condition index values measured at Pitt Water and Pipeclay Lagoon, indicate lower growth and greater diversion of energy into shell growth and metabolic maintenance, with reduced growth of soft body mass. Aldrich and Crowley (1986) in a study of intertidal and raft cultured mussels, found low meat weights and higher shell weights in intertidal mussels, and high meat weights and low shell weights of raft cultured mussels. These authors attributed the differences between the two to the slow growth of intertidal mussels and poor environmental conditions.

Condition measured using the Crosby and Gale (1990) index, similarly showed lower values at Pitt Water and Pipeclay Lagoon, with the exception of higher values at Pitt Water on the third trial (site 1) and Pipeclay Lagoon in the second and third trials. This was attributed to the greater increase in meat weight relative to live weight. However, often high values were attributed to low shell cavity capacity, measured as the difference between live weight and dry shell weight. Thus, at times oysters with large shell weights relative to total weight, and hence low shell cavity capacity, resulted in deceptively high Crosby and Gale condition indices. This accounts for the large variation shown in the Pitt Water oysters at site 1 at the end of the first three months. Interestingly, the oysters from site 2 in trial 1 at Pitt Water, which had shown little change in shell length, were in good condition as indicated by the average condition indices. Maguire et al. (1994) state that the minimum acceptable value using this condition index is ≥ 70 for market product of Pacific oysters in Tasmania.

Both Lucas and Beninger and Crosby and Gale condition indices were much higher at Little Swanport, and indicated more favourable growth conditions. Lower mean Lucas

and Beninger condition indices, compared to the initial values, were measured at the end of trials 1 and 2, with an increase shown at the end of trial 3 attributed to much larger percentage increase in dry meat weights relative to dry shell weights.

Often the trend in the two types of condition indices used differed, such as higher Crosby and Gale CI but lower Lucas and Beninger CI values. One reason for this has been described above. Overall, the Lucas and Beninger method was considered to yield the more reliable and representative results of oyster condition.

The comparisons made between each area and trials using mean instantaneous daily growth rates, showed Little Swanport to generally have the higher values for shell length, whole and dry meat weight growth. However, Pitt Water showed the highest mean instantaneous daily dry meat weight growth rates at both sites at the end of trial 2 (summer period), indicating that environmental conditions at this time were highly favourable for soft body growth.

Few studies of oyster growth rates have been conducted in Tasmania by which comparisons can be made, and none have been conducted assessing water quality parameters with the exception of temperature and salinity. Hallegraeff et al. (1988) conducted growth studies in Pitt Water using similar methods to those in this study. These authors assessed growth of oysters of similar initial size and density over two six month periods from August 1985-January 1986 and February 1986-July 1986. Percentage increases at the end of each trial were 30% and 45% for shell length, 26% and 51% for width and 26% and 48% for depth (Hallegraeff et al., 1988). No information was given on live weights. Whilst the time of year differed slightly from the trials conducted in this study, it can be seen that shell growth was much greater in the 1985/86 trials than the present ones. It is difficult to assess the probable reason for these differences, which may be due to subsequent increased stock density and lease areas within Pitt Water, altered environmental conditions (e.g. the Craigbourne Dam construction), genetic variability between parent hatchery stock for the spat used, or other reasons. Nevertheless, changes in oyster growth rates have occurred over this time, and this has been an observation noted also by the oyster farmers.

A study conducted by Maguire et al. (1994) on comparison between diploid and triploid oysters at Pitt Water and Little Swanport, showed growth of diploid oysters to be faster at both sites, with greater glycogen content in the oysters at Little Swanport. These authors found higher rates of growth at Little Swanport than at Pitt Water (>60g/oyster

at 19.9 months compared to 22.5 months at Pitt Water). Over the duration of their trial, no spawning was noted at Little Swanport, but spawning did occur at Pitt Water. Oyster condition and meat weight declined at Little Swanport during this time, with a relatively rapid increase in recovery shown after this time. However, dry meat weight and condition of the oysters at Pitt Water post-spawning remained low, with a slow rate of recovery until approximately four months later (Maguire et al., 1994). Regular sampling was done at approximately 3 month intervals for the duration of their trial, with little apparent seasonal growth pattern shown. Growth rates were also found to be influenced by exposure time, with greater shell growth shown at much shorter periods of exposure at Pitt Water (10.9% exposure) compared to Little Swanport (59.2% exposure) (Maguire et al., 1994).

Sumner (1980) studied the growth of wild caught oyster spat from the Tamar River, which were on-grown on a lease in Pipeclay Lagoon during 1971/73. Percentage increase in shell length over the first 12 months was 67% (final mean length 75mm), with seasonal differences in growth found. Greater shell increases occurred over autumn, with preferential increase in width over length during winter. Reduced growth rates were found during the summer period, which Sumner (1980) attributed to a combination of high air temperatures (oysters were set at 40% exposure level on racking) and reproductive activity. Similar observation of preferential increase in width of oysters at Pipeclay Lagoon were noted during the winter period (trials 1 and 3) in this study.

In the present study, the oysters from Little Swanport had lower initial shell weights, as compared to the initial weight of oysters used at Pitt Water and Pipeclay Lagoon. It is most likely that the Little Swanport oysters were younger than those used at the other sites. Size and age could have influenced the growth rates shown. Faster growth in juvenile and small oysters have been noted elsewhere (e.g. Cigarría, 1999; Mitchell et al., 2000). Change in shell length was compared to mean initial shell length of oysters at each site in the three study areas (Fig. 5.17). For comparison, the shell length data for the 1985/85 Pitt Water growth study (Hallegraeff et al., 1988) have been included. It can be seen from Fig. 5.17, that there is an apparent trend of change in shell length with initial shell size. However, some variation was shown, particularly at Pitt Water at the end of trials 1 and 3. It appears that other factors are most likely responsible for change in shell length.

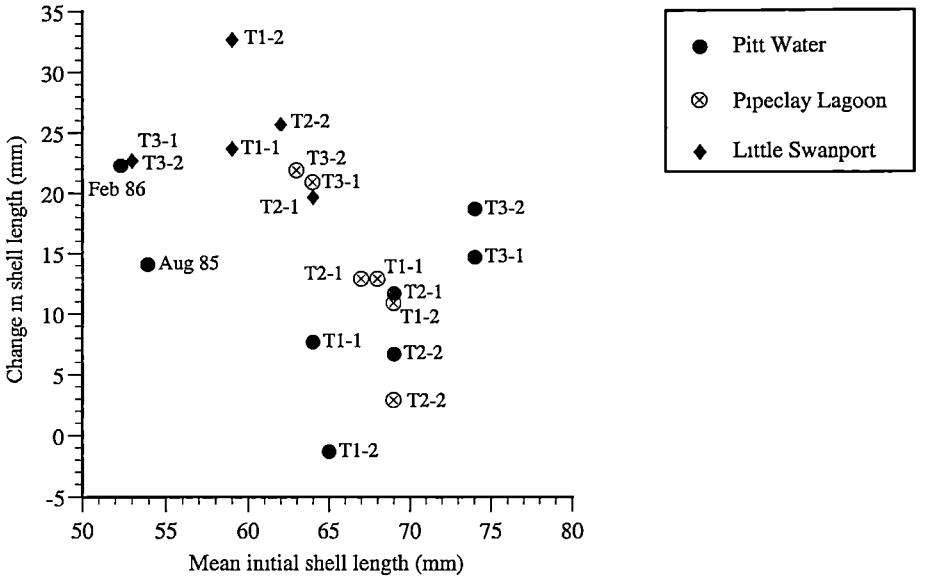

Fig. 5.17 Change in shell length for two sites (-1, -2) at Pitt Water, Pipeclay Lagoon and Little Swanport for three trial periods (T1-, T2-, T3-). Aug 85 and Feb 86 Pitt Water data included (ref to text).

Ideally, for better assessment of growth and comparison between the three study areas, oysters should have been supplied from the same source and distributed to each area. This would have ensured greater control over the initial sizes used, and minimised any possible influences due to genetic variability. More frequent and consistent measurements (with respect to initial start and end dates of trials) would have enabled greater assessment of trends in growth and condition. However, this requires a considerable amount of time and effort, which unfortunately was not possible to undertake in this study. Measurements on individually tagged oysters have been used to provide a means of calculating growth curves for more accurate assessment of growth rates (Mitchell et al., 2000). The ability to generate growth curves also provide a useful means of predicting the time required to reach a specified market length or live weight.

6. General Discussion

This study has provided a progressive understanding of likely factors responsible for differences in oyster growth rates and shellfish productivity reported among the three areas assessed. The data provides a greater insight into food quality and quantity available to the farmed oyster populations, differences between the growing areas, and the hydrodynamic influences on the source and supply of this material.

On the basis of information available from oyster grower's production returns for the period 1995/96, Pipeclay Lagoon had the highest oyster production per hectare of lease area, with approximately three times more oysters produced/ha than at Pitt Water (DPIF, 1998; Crawford and Mitchell, 1999). The reported 1995/96 oyster production figures were 17 x 10^4 oysters/ha for Pipeclay Lagoon, 5.7 x 10^4 oysters/ha for Pitt Water and 5 x 10^4 oysters/ha for Little Swanport. However, the present study showed oyster growth rates were higher at Little Swanport. Pipeclay Lagoon showed lower values to those calculated at Little Swanport, and Pitt Water generally had the lowest. Considerable differences in the time taken for seed oysters to reach market size were observed. Approximate grow-out times, as reported by oyster farmers (pers. comm.), are 18 months in Little Swanport, 36 months in Pipeclay Lagoon and 42 months in Pitt Water.

Most of the existing lease areas in Pipeclay Lagoon are fully developed with the exception of two leases; one lease is approximately 70% developed and the other has 2.6 ha which is too shallow to farm (DPIF, 1998). It appears that oyster production from the current lease areas in Pipeclay Lagoon is near optimum levels, with respect to growing times and production (DPIF, 1998). A recent Marine Farming Development Plan for the lagoon proposed an increase in the marine farming zone areas, but no increase in leaseable area (DPIF, 1998). This was done to enable greater spacing between leases, prevent placement of racking within flow channels and to improve water flow through the leases.

Oyster farmers in Pitt Water indicate there are problems with oyster production and growth rates despite the relatively large total area of leases. Not all leases are fully developed, but no figures are available as to the approximate size of developed area. Crawford and Mitchell (1999) reported concerns raised by the oyster farmers in this area of the changes to the dynamics of the lower estuary, as a consequence of the

Craigbourne Dam in the upper reaches of the Coal River. This dam has altered flows and inputs of freshwater to the lower estuary, where the shellfish farms are located. The concerns raised were specifically related to the effects of altered flow patterns on reducing nutrient inputs and primary productivity, and hence reducing growth rates and production on leases.

The majority of lease areas in Little Swanport have been developed, with the exception of the lease located in the mid-region of the estuary which is partially developed. Good oyster growth rates and condition are experienced in this area, with reported production rates per ha of lease marginally lower than Pitt Water. However, growth rates are much faster and indicate favourable conditions at the level of production reported. It is difficult to assess if, and to what level, production could be increased whilst still maintaining this growth rate.

In order to gain some understanding of conditions within each of these areas, an assessment was made of those water quality parameters recognised as being influential to shell fish growth and productivity (Brown and Hartwick, 1988a; Carver and Mallet, 1990; Roland and Brown, 1990; Hickman et al., 1991; Ball et al., 1997; Soniat et al., 1998; Toro et al., 1999). These included temperature, salinity, TPM, POM,PIM and phytoplankton abundance. Nutrients (NOX-N, PO_4-P and SiO_4-Si) were also measured to determine correlation with phytoplankton abundance.

Pitt Water was the largest of the three major oyster growing areas studied. Total length of the estuary was approximately 21 km, with a surface area approximately 10 times that of Pipeclay Lagoon and seven times greater than Little Swanport. A causeway located approximately mid-estuary, effectively divides the estuary into two regions referred to as Upper and Lower Pitt Water. The connection between these two regions is via the narrow opening of the causeway bridge, a gap width of approximately 480 m. Each of the three oyster growing areas studied, was characterised as having extensive regions of shallow water with a relatively narrow deeper channel leading from the mouth to the upper reaches. A common feature of each was a narrow mouth where tidal exchange occurs, this being almost identical in width at each area.

A seasonal trend in temperature fluctuations occurred in each area, with lowest temperatures recorded in winter and highest in summer. However, the greater extremes of temperature were recorded in Pitt Water (4.4-23.8°C) at a shallow site in the upper estuary. Salinity was often higher within Pitt Water and Pipeclay Lagoon, compared to

the marine sites, during the warmer months, suggesting the effects of evaporation. Similar observations have previously been noted in these areas (Crawford and Mitchell, 1999). Salinities were considerably depressed following rainfall events in the later part of 1995 to early 1996 at Pitt Water and Little Swanport. Pipeclay Lagoon showed minimal change in salinities during this same time period, despite exposure to similar rainfall intensities. Reductions in salinity appeared to be influenced by previous rainfall history and presumably soil saturation levels in the catchment. This was indicated by the 7 and 12 day accumulative rainfall data previous to the sampling date.

Assessment of seston (food) quality and quantity over the 13 month period showed considerable variation among areas. A seasonal trend was evident in phytoplankton abundance, with chlorophyll a levels declining in late summer to autumn, and high levels at all sites in late winter-early spring. A decline was noted at Pitt Water marine site in summer but not at the estuary sites, most likely as a consequence of the freshwater inputs. Chlorophyll a levels were often higher within the estuary or coastal embayment sites compared to the marine site, suggesting either autochthonous phytoplankton production or inputs from other sources (e.g. benthic phytoplankton, seagrass debris). A gradient of increasing chlorophyll a towards the upper reaches was noted in Pitt Water and Little Swanport.

Overall comparison between each area, based on the mean values calculated from all sites within Little Swanport, Pipeclay Lagoon and Pitt Water, which was divided into two sections; Upper (sites 4-7) and Lower (sites 2 and 3) are shown (Fig. 6.1). The marine sites were not included in this analysis, and are shown separately (Fig. 6.2). The notable features from Fig. 6.1 show the highest mean TPM levels were in Upper Pitt Water, and lowest in Little Swanport. POM concentrations were similar in Lower Pitt Water and Little Swanport, but lower than those recorded in Upper Pitt Water and Pipeclay Lagoon. However, %POM was highest at Little Swanport, followed by Pipeclay Lagoon with Pitt Water showing the lower values, particularly Upper Pitt Water. At Pitt Water, this was due to the higher inorganic fraction of the suspended particulate matter evident in this region.

Mean chlorophyll a concentrations were similar in Upper Pitt Water and Pipeclay Lagoon, though greater variation was shown at Pipeclay Lagoon. Little Swanport recorded the higher levels and similarly showed greater variation that at Pitt Water. In

Pipeclay Lagoon, a similar seasonal trend in chlorophyll a and %POM indicated a close association of %POM with phytoplankton abundance.

NOX and PO_4-P were low, with the higher NOX concentrations recorded in Little Swanport followed by Pipeclay Lagoon. Mean PO_4-P concentrations were similar in Lower Pitt Water and Pipeclay Lagoon, with the lowest levels in Little Swanport. A trend of decreasing PO_4-P concentrations progressing up the estuary was noted in Little Swanport. Often concentrations of NOX and PO_4-P were higher at sites within the estuary and coastal lagoon areas compared to their respective marine sites, with the exception of NOX peaks at the marine site during winter at Pipeclay Lagoon and Little Swanport, and Pipeclay Lagoon in November 1995. SiO_4-Si concentrations were considerably lower in Pipeclay Lagoon than those measured in Pitt Water and Little Swanport. The latter two sites showed a trend of increasing levels progressing up the estuary. Higher levels were recorded in these areas following freshwater inflows and depressed salinities.

NOX concentrations never fell below non-detectable levels in each area. It has been noted that low NOX levels do not necessarily imply nitrogen limitation, but perhaps rapid utilisation and recycling may be occurring (Day, 1981b; Koop et al., 1998). As mentioned above, on occasions higher chlorophyll a levels were measured within the estuary/coastal embayment region of each area. Whilst these may have been attributed to the sources suggested, the possibility exists that remineralization of nitrogen, most likely by nitrification, may have been occurring. At times these measurements coincided with moderate NOX concentrations, attributed to freshwater inputs of NOX, whilst at other times remineralization appeared to be occurring. Similar observations and assumptions have been noted elsewhere (Sornin et al., 1990; Gibbs et al., 1992; Ball et al., 1997). The rate and amount of nitrogen regeneration within each system could have been estimated from concurrent measurements of ammonium concentrations (Dugdale and Goering, 1967).

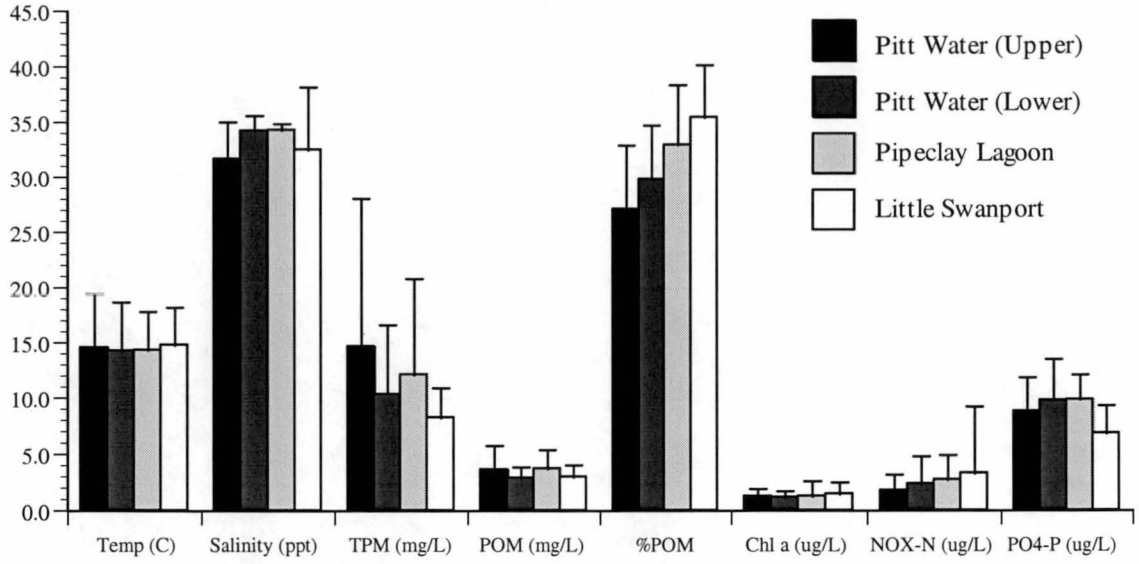

Fig. 6.1 Mean (± sd) temperature (C), salinity (‰), total particulate matter (TPM) and particulate organic matter (POM) - mg/L, %POM, chlorophyll a (μg/L), NOX-N (μg/L) and PO_4-P (μg/L) at Pitt Water Upper and Lower estuary, Pipeclay Lagoon and Little Swanport.

Comparison of the marine stations show a similar trend of %POM to above, in that both Pipeclay Lagoon and Little Swanport recorded the higher levels (Fig. 6.2). Mean TPM and POM concentrations were similar at Pitt Water and Pipeclay Lagoon, with mean chlorophyll a concentrations higher at Pipeclay Lagoon and Little Swanport. The most notable difference is shown in the NOX concentrations with the highest levels recorded at Little Swanport during the colder months. This pattern was also noted at Pipeclay Lagoon. This has been linked to intrusions of nutrient rich water masses with associated elevated phytoplankton biomass during this period of the year, as recorded at Maria Island on the east coast (Harris et al., 1987) and Storm Bay (Clementson et al., 1989; Harris et al., 1991).

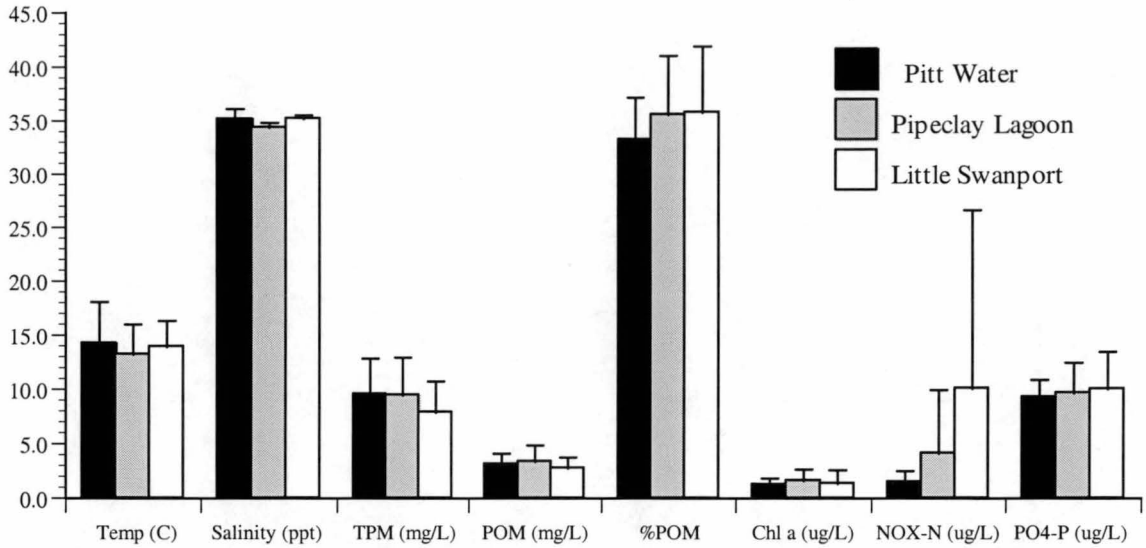

Fig. 6.2 Mean (± sd) temperature (C), salinity (‰), total particulate matter (TPM) and particulate organic matter (POM) - mg/L, %POM, chlorophyll a (μg/L), NOX-N and PO_4-P of marine station at Pitt Water, Pipeclay Lagoon and Little Swanport.

Oyster growth rates, measured by increase in shell length, live weight, width and depth were greatest at Little Swanport. Condition indices were also generally higher than at Pitt Water and Pipeclay Lagoon, indicative of more favourable growth conditions. Significant increase in dry meat weight was shown in the colder (winter) time of the year, and lower increase over the warmer period at Little Swanport. However, considerable increases in shell length and live weight were shown at the end of each trial. Chlorophyll a levels within the estuary were relatively high throughout most of trials 1 and 2, and never fell below 0.4 $\mu g\ L^{-1}$. POM levels over this same time were within the range 2-4 mg L^{-1}.

In Pitt Water and Pipeclay Lagoon, little percentage increase in shell length and live weight were recorded during the winter period (trial 1). However, the greatest mean instantaneous daily dry meat weight growth was shown at Pitt Water at the end of trial 2, which coincided with the freshwater inflows and greater increase in chlorophyll a and POM levels prior to this time. Greater increase in live weight compared to shell length was noted during this summer period, and reflected favourable conditions for soft body tissue growth. Final measurements at the end of trial 1 coincided with the time of reduced chlorophyll a levels, with the lower increase in dry meat weight possibly reflecting the drawing of glycogen reserves by the oysters at that time (Quayle, 1969) due to reduced seston availability. In Pitt Water and Pipeclay Lagoon, live weight increases were shown to be due to greater deposition of shell rather than soft body

weight, suggesting a reduced capacity for tissue growth with food intake largely directed towards metabolic maintenance.

Measurement of glycogen content has often been used to assess correlation of oyster soft body growth with food availability (e.g. Deslous-Paoli and Héral, 1988; Almeida et al., 1997; Almeida et al., 1999). Maguire et al. (1994) measured higher glycogen content in oysters grown in Little Swanport compared to Pitt Water. Glycogen content at both sites declined to minimum values during the second summer of their trial prior to spawning, with a steady increase in levels after this time shown at Little Swanport. During their study, spawning of the oysters occurred in Pitt Water but not at Little Swanport. Dry meat weights of the Little Swanport oysters increased after this time, but at Pitt Water little change in dry meat weights were noted for a period of approximately four months post-spawning. Interestingly, at Little Swanport, greater increase was shown in dry meat weights with reduced shell weight growth, whereas the opposite was shown in Pitt Water. Comparison of glycogen contents by Maguire et al. (1994) with the chlorophyll a levels measured by Crawford et al. (1996) over the same time period, indicate a relationship with chlorophyll a levels. Higher chlorophyll a levels were recorded in Little Swanport over the four month period and lower declining levels were recorded at Pitt Water. This indicates that environmental conditions at that time, most likely reduced food availability, reduced the development and recovery of soft body mass in the oysters at Pitt Water.

Deslous-Paoli and Heral (1988) observed a seasonal cycle of storage and utilisation of glycogen, which was closely linked to oyster growth and the reproductive cycle. These authors noted a strong link in glycogen content with the quality and quantity of food available during phytoplankton blooms in spring and autumn. Almeida et al. (1999) also reported a similar pattern of glycogen storage and transformation, which was closely linked to phytoplankton composition. The two sites which these authors studied showed similar environmental conditions, however differences were apparent in the biochemical composition of the oysters, attributed to differences in phytoplankton composition.

A study of phytoplankton composition in Pipeclay Lagoon and Little Swanport by Brown and McCausland (1999) showed concentrations of long-chain polyunsaturated fatty acids (PUFAs) at Little Swanport to be approximately double those in Pipeclay Lagoon, reflective of the higher and richer phytoplankton community. Additionally, van

den Enden (1994) showed that oysters in Little Swanport selectively fed on specific phytoplankton species, with benthic diatoms and seagrass detritus a significant component of the oysters diet. He also indicated that oysters altered their feeding rate in response to food availability in the water column. Similar observations of selective feeding and altered feeding rates have been noted in Pitt Water (Hallegraeff et al., 1986). The marine source for both Pitt Water and Pipeclay Lagoon is Frederick Henry and Storm Bay, however no inferences can be made between the two with respect to phytoplankton composition and quality. Hallegraeff et al. (1986) indicated that the phytoplankton composition in Upper Pitt Water was quite distinct from Storm Bay and predominantly composed of benthic diatom species originating from within the estuary.

POM is frequently used as a measure of available food for bivalves (Carver and Mallet, 1990; Hickman et al., 1991; Ball et al.,1997; Hawkins et al., 1998; Grant and Bacher, 1998). However, consideration needs to be given to the quality of this fraction with respect to labile and refractory composition. The general method used for determination of this component in seston (i.e. ashed filter papers), does not provide an estimate of the labile, readily utilisable organic fraction. Similarly, Ball et al. (1997) indicated that such measures may overestimate the labile, biologically available organic matter. These authors showed high levels of particulate organic carbon (POC) in winter corresponded to low labile POC levels associated with the resuspension of more refractory organic matter. No trend was apparent in POM levels in each of the three areas during the present study. On occasions high chlorophyll a levels coincided with high POM levels, whilst at other times detrital matter appeared to be a major component. However, as mentioned above, the organic quality of this fraction was unknown.

Turbidity, or high TPM, especially high PIM levels, have been shown to have a deleterious influence on filtration and hence growth rate in oysters (e.g. Barillé et al., 1997). A reduction in filtration rates from 5 L $h^{-1}ind^{-1}$ to 3.5 L $h^{-1}ind^{-1}$ in seston range of 60 - 192 mg L^{-1}, with filtration ceasing at levels >192 mg L^{-1} has been shown in Marennes-Oléron Bay (Barillé et al., 1997). This bay has been characterised as having a high TPM load, fluctuating from 20 - 350 mg L^{-1} due to tidal and wind driven resuspension, with the PIM varying from 65-95% (Barillé and Prou, 1993; Barillé et al., 1997; Grant and Bacher, 1998). It has been shown that *Crassostrea gigas* feed selectively, enriching the organic fraction ingested and rejecting inorganic or less nutritional matter as pseudofaeces (e.g. Pastoureaud et al., 1996; Hawkins et al., 1998). Hawkins et al. (1998) found that pseudofaeces production does not necessarily occur

above a threshold value of TPM, indicating that selectivity of ingested matter occurs even at low TPM concentrations. Clearance rates and selection efficiency (or capacity), vary according to seston loads, with a reduction shown at high loads (> 50 mg L^{-1}) attributed to saturation of ingestive or processing capacity (Pastoureaud et al., 1996), or exceeding the clogging threshold at high loads (>192 mg L^{-1}) (Barillé and Prou, 1993).

TPM concentrations at Little Swanport were less than 20 mg L^{-1} and less than 40 mg L^{-1} at Pitt Water and Pipeclay Lagoon, with the exception of moderately higher values attributed to the sample bottle intake line being close to the sediment surface. Whilst the water in Upper Pitt Water was frequently observed to be turbid, due to suspension of silt/clays, the levels of TPM measured are low in comparison to Marennes-Oléron Bay.

The oyster growth rates and pattern of growth reflect the influence of environmental factors in each area. As previously indicated, oyster growth is influenced by temperature, salinity and available food (e.g. Brown and Hartwick, 1988a, 1988b; Roland and Brown, 1990), with notable differences in shell lengths, live weights and dry meat weights particularly dependent on available food. Brown and Hartwick (1988a) used a classification of high, medium and low growth sites based on temperature, phytoplankton biomass (chlorophyll a) and salinity. Oysters at high growth sites displayed greater increases in shell length, whole oyster, shell and dry meat weights, which were attributed to high chlorophyll a levels and suitable temperature and salinity. Medium growth sites displayed significantly lower growth in shell lengths and dry meat weights, but whole oyster and shell weight growth was equivalent to high growth sites. The notable difference between the two site classifications were lower chlorophyll a levels. Low growth sites showed reduced growth, principally attributed to high chlorophyll a levels coinciding with suboptimal salinities (<20‰). Using this classification, and based on the results presented in Fig. 6.1 (in particular chlorophyll a, POM and %POM), Little Swanport could be regarded as a high growth site, whereas Pipeclay Lagoon and Pitt Water could be considered medium growth sites.

Brown and Hartwick (1988b) indicated that shell thickening was partially related to food availability, i.e. low chlorophyll a, as under conditions of prolonged low food supply, shell growth and maintenance is more energetically efficient than soft tissue growth. They also noted that oysters at poor growth sites can have similar condition index values to those recorded at high growth sites. The rate of recovery of body tissue weights following spawning is influenced by availability of food (Almeida et al., 1999)

and similarly, food availability influences the energy requirements for metabolic maintenance and hence growth (e.g. Malouf and Breese, 1977; Deslous-Paoli and Heral, 1988). Thus, under these conditions, the amount of food available and ingested is sufficient to maintain metabolic functioning, but insufficient to permit greater storage of reserves and hence increase in soft tissue weight. This appears to be the likely process occurring in Pitt Water and Pipeclay Lagoon. However, the greater phytoplankton biomass recorded following the freshwater inflows and reduced salinities in the Pitt Water estuary resulted in greater soft tissue growth as reflected in the higher dry meat weights recorded at that time.

An important factor in oyster growth rate and condition is the rate of supply of food to the oysters, namely consideration of the hydrodynamic characteristics within an area. Water flow, and hence food supply, has been recognised as an important factor in bivalve culture operations with a number of studies showing that the supply and transport of food across bivalve culture structures, or native beds, can lead to depletion of food to subsequent downstream populations (Rosenberg and Loo, 1983; Dame and Dankers, 1988; Navarro et al., 1991; Dame et al., 1992; Heasman et al., 1998). Considerable variation in growth rate can occur as a consequence, leading to considerable influence on the productivity and carrying capacity of culture systems (e.g. Smaal and Prins, 1993).

Following a review of a number of bivalve systems, Smaal and Prins (1993) suggested that the scale of impact of bivalve culture, and hence carrying capacity, could be determined from comparison of the residence time and clearance rate of the bivalve population, i.e. the time taken for the population to filter a volume equivalent to the system volume. Using biomass expressed as g/m^3 of the total volume, these authors showed that when the biomass/unit volume is low, the impact of filtration of the bivalves is limited to the scale of the population and the impact at the ecosystem level would be small. However, if the biomass/unit volume is large, clearance times were often shorter than the residence times and the impact at the ecosystem level could be potentially significant. The level at which impacts were noted was when the biomass/unit volume was in the range of 2-8 g ash-free dry weight (AFDW). This approach was used in a study of Carlingford Lough, Ireland (Ball et al., 1997; Ferreira et al., 1998) and has been used here. The table from Smaal and Prins (1993) is shown below (Table 6.1) with additions calculated from studies conducted elsewhere as indicated. The model used for calculating the values tabulated for Pitt Water, Pipeclay

Lagoon and Little Swanport is described in Appendix 4.1. It must be noted that the values calculated were based on approximations of standing stocks within each area. Actual figures were difficult to obtain principally for two reasons: details of production and stocking density obtained from oyster farmers returns is regarded as confidential information; and secondly, standing stock is often difficult to assess due to movement of stock as some farms grow oysters to a certain size (e.g. 50 mm) which are then moved for on-growing elsewhere (generally at a different location within the state), or oysters are brought in from elsewhere for on-growing to harvest.

The results of this assessment indicate that the scale of impact of the culture areas in Pitt Water are confined to the lease areas, as the clearance time is considerably greater than the residence time calculated for Upper Pitt Water. This is also shown at Little Swanport, however, growth rates currently found within this region do not support the assumption that significant impact is occurring on the culture system. Pipeclay Lagoon could be regarded as reaching the upper limit of production from this system, as the clearance times are comparable to the residence time. This conclusion is supported by the results of the oyster growth trials.

It is interesting to compare the culture systems studied in Tasmania with those overseas. With the exception of North Inlet, Sylt and Nova Scotia (Table 6.1), the scale of the Tasmanian areas, in terms of water volume, are considerably smaller than those in France, Spain, Ireland and the Netherlands (see Table 6.1). Standing stock is also generally considerably less. A review of the culture systems reported in this table show notable differences between the north and southern hemisphere. For example, culture areas in France, Ireland and the Netherlands are macrotidal with relatively high flushing and cxchange rates. As previously stated, TPM within Marennes-Oléron is highly variable (e.g. Smaal and Zurburg, 1997; Grant and Bacher, 1998). Similarly, chlorophyll a and POM concentrations are also highly variable, with values measured within the approximate range 0.1 - 18 ug L^{-1} and 0.5 - 20 mg L^{-1}, respectively (Grant and Bacher, 1998). *Crassostrea gigas* growth times currently reported in the bay are approximately 3 - 4 years (Raillard and Ménesguen, 1994; Ferreira et al., 1998), which is comparable to those in Pitt Water and Pipeclay Lagoon.

Table 6.1 Adapted from Smaal and Prins (1993). Total system volume, mean biomass (as AFDW (unless otherwise indicated) per m^2 of total surface and per m^3 of total volume), standing stock, daily clearance rate (CL) and clearance time in days in comparison to residence times.

Area	Total Vol. 10^6 m^3	Average depth (m)	Biomass g m^{-2}	Biomass g m^{-3}	Standing stock 10^6 g	CL Rate L $g^{-1}day^{-1}$	CL Rate m^3 $m^{-2}day^{-1}$	CL Time day	Residence time (days)
South San Fransisco Bay	2500	6	15	2.5	6255	600	9	0.7	11.1
North Inlet, South Carolina (USA)	22 [a]	2.5	10.6 [c]	15.4	338	86	3.2	0.8	1
Sylt	7.2	1.8	39	21.7	156	22.5	0.9	2.1	0.5
Asko (Baltic Sea)	4000	25	9.4 [c]	0.4	1500	26.9	0.3	99	10000
West Wadden Sea (NL)	4020	2.9	10.4	3.1	14700	48	0.5	5.7	10
Oosterschelde (NL)	2740	7.8	24.3	3.1	8509	87	2.1	3.7	40
* Eastern Scheldt (NL) [1]	2740		13.2	1.67	4580	48	0.5	12.5	20-135 [d]
Brest (France)	1480	10	74	7.4	10952	48	3.5	2.8	16.7
Marennes-Oleron (France)	675	5	21	4.2	2850	86.4	1.8	2.7	7.1
Thau Lagoon (France)	265	5	15.9	3	800	120	1.8	2.8	
Ria de Arosa (Spain)	4335	19	30	1.6	6900	51	1.5	12.4	23
Killary Harbour (Ireland)	102	14	2.1	0.2	15.5	48	0.1	140	
* Carlingford Lough (Ireland) [2]	385	5		0.1		50			
* Nova Scotia (Canada) [3]	1.8	3.6		1.66	2.99	46.8		12.8	
* Pitt Water (Upper) - this study	41.2 [b]	2	0.5 [c]	0.28	11.4	50	0.04	72.2	7-17 [d]
* Pipeclay Lagoon - this study	3.5 [b]	1	2.8 [c]	3.64	12.7	50	0.2	5.5	2.05-6.85 [d]
* Little Swanport - this study	6.6 [b]	1	0.8 [c]	0.75	4.9	50	0.05	26.6	1.97-4.26 [d]

* additions to table
a: tidal prism; b: mean volume; c: biomass in dry flesh weight; d: range in residence times.

1: Eastern Scheldt (NL), mussels, after Dame et al (1991); 2: Carlingford Lough (IF), mainly oysters, after Ball et al. (1997) and Ferreira et al. (1998); 3: Nova Scotia (C), mussels, after Carver and Mallet (1990).

Microphytobenthos and resuspended biodeposits have been shown to form a significant component of oyster diets in Marennes-Oléron (Pastoureaud et al., 1996). Chlorophyll a levels in the North Inlet (Dame and Libes, 1993), Western Wadden Sea and Eastern Scheldt (Dame et al., 1991) are considerably higher in comparison to those measured in this study. Carlingford Lough (Ireland) is approximately 16.5 km in length (Ball et al., 1997) and shorter than Pitt Water, however water volume, exchange rates, nutrients and seston levels are considerably greater than those measured in Tasmania. Chlorophyll a and TPM levels were generally in the range 1.5 - 6 mg m^{-3} and 5 - 66 g m^{-3}, respectively (Ball et al., 1997). Oyster growth rates in Carlingford Lough are comparable to those in Little Swanport (~ 1.5 years), and an ecosystem box model used to assess the carrying capacity of this system, calculated the carrying capacity to be 0.44 g AFDW/m^3 or 0.26 oysters/m^3 (Ferreira et al., 1998).

The results of the hydrodynamic study are in close agreement with the observations noted within each area. Briefly, the marine nature of Pipeclay Lagoon is shown by the relatively short residence times, large percentage mean tidal prism and high exchange rates. Additionally, of interest, the low silicon levels also indicate a more marine nature of this system. Calculated mean residence times in Little Swanport were similar to Pipeclay Lagoon, though the mean percentage tidal prism and exchange rates were slightly less. This estuary was subject to freshwater inflows and reductions in salinity, but showed a relatively rapid return to normal salinities. Pitt Water showed the lower mean tidal prisms and exchange ratios in both the Upper and Lower sectors, and longer residence times. The more appropriate estimate of mean residence time in Upper Pitt Water was considered to be 7-17 days. Similarly to Little Swanport, this estuary was subject to flooding with reductions in salinity, but showed a slower return to normal salinity levels after these events. Considerable differences were shown in the mean flow rates within the culture region of each system. Little Swanport flows were faster than those calculated for Pipeclay Lagoon and Upper Pitt Water; 0.14, 0.07 and 0.04 m s^{-1}, respectively.

It appears that flow rates within the culture sites play a significant role in the observed oyster growth rates. Boyd and Heasman (1998) showed flow rates within and between mussel rafts in Saldanha Bay to be highly variable, with retardation of flow by the mussel ropes significantly influencing the rate of food supply and hence production. Grant et al. (1998) in a study of the same bay, used passage of water as a measure of food delivery, as estimated from the transit time of a particle through a 1 ha body of

water. Restriction, or retardation, of flow due to the mussel ropes was estimated to cause a 6 fold decrease in current speed. These authors calculated the turnover of water per day, assuming 24 hrs of flow, as 24 ÷ transit time. Adopting a similar approach using the mean flows for each area, the transit time of a particle across a 1 ha section of lease with unimpeded flow, 2 and 4 fold retardation in flow due to rack structures and the corresponding turnover times were calculated (Table 6.2). No figures were found on actual values for retardation of flow due to rack structures, hence the arbitrary use of 2 and 4 fold impedance values.

Table 6.2 Food delivery rates as estimated by particle transit time across a 1 ha section of lease using 0, 2 and 4 fold impedance to flow, with corresponding number of turnovers of water per day.

		Particle transit time (hr)			Turnovers (per day)		
Area	Mean flow	0	2	4	0	2	4
Upper Pitt Water	0.04 m s^{-1}	0.69	1.39	2.78	34.8	17.3	8.6
Pipeclay Lagoon	0.07 m s^{-1}	0.40	0.79	1.59	60.0	30.3	15.1
Little Swanport	0.14 m s^{-1}	0.20	0.39	0.79	120.0	61.5	30.4

These figures provide some estimation of the rate of food supply within the respective culture systems, and hence the likely consequences of this with respect to food availability and thus growth rate of oysters. These estimates have been calculated using mean flow rates, though it is recognised that considerable variation occurs in flow rates during tidal cycles, as shown in the tidal flux study at Pipeclay Lagoon. Flow rates also vary depending on the difference between high and low water heights, greater flows occurring when the differences are larger.

It has been shown that oysters to a large extent can regulate their filtration rate. However, successive depletion invariably occurs across culture systems unless they are widely spaced apart (Grant, 1996). Oysters on racking closest to the incoming flow of water selectively remove particulate matter from the water column, expelling excess, or less desirable, matter as pseudofaeces. This results in a reduction of desirable food to successive rows normal to the incoming flow. This is exacerbated if flow rates are reduced, permitting leading rows more time to selectively filter the incoming water. This concept of flow rate and bivalve growth has been illustrated by Wildish and Kristmanson (1997), whereby increases in flow permit greater seston concentration availability to downstream populations. Simplistically, a greater amount of seston can pass the leading oysters unaltered without being filtered and bound in pseudofaeces.

Resuspension, or successive transport of biodeposits can provide some means of increasing food supply to subsequent rows, however the nutritive quality of this matter is dependent on the quality of the initial source. Crawford and Mitchell (1999) showed that oysters in Pitt Water and Pipeclay Lagoon increased the inorganic fraction of biodeposits, indicating preferential selection of organic matter. This was also shown in Pipeclay Lagoon from the results of the biodeposition study (this study). However, the organic fraction of this material was considerably reduced, and hence the organic quality to subsequent rows would be considerably reduced. Studies would need to be conducted to ascertain the length of lease area in which this successive depletion in seston quality and quantity (inclusive of contributions from biodeposits) would become critical to oyster metabolic maintenance and growth. Intensive sampling conducted around leases in each of these areas during high and low tides on occasion did indicate depletion of chlorophyll a across lease areas (Crawford and Mitchell, 1999). However, more intensive sampling upstream and downstream of lease areas would be required to confirm this observation and quantify the level of depletion.

The results of the biodeposition study indicate that biodeposits could contribute a significant component of oyster diets, for example via resuspension or transport to subsequent oysters. In light of studies conducted elsewhere, they could also assist with enhancement of benthic phytoplankton biomass (e.g. Guarini et al., 1998) and nutrient recycling following mineralization (e.g. Sornin et al., 1983; Barranguet et al., 1994; Grant et al., 1995).

In the course of this study, a number of questions became apparent in order to try and fully explain and understand the dynamics and processes occurring in each area. These ranged from assessing the dispersion of biodeposits and fate of this material on changes in microbial and benthic community structure, assessment of benthic nutrient fluxes and hence the degree of mineralisation and autochthonous nutrient sources, more intense assessment of food quality and sources within each area, and *in situ* studies on filtration, clearance rates and seston depletion.

The development of a carrying capacity model appropriate to local conditions, with particular emphasis on system hydrodynamic characteristics, is seen as providing a means of enabling greater understanding and prediction of appropriate flow rates within culture areas. This would also provide a means of estimating appropriate spatial arrangement within, and between, culture leases. This would provide both

environmental mangers and oyster farmers a valuable tool with which to maximise shellfish productivity in a given region.

This study raised the question that within Pipeclay Lagoon, biodeposits appeared not to be accumulating under the shellfish culture structures, but were most likely being transported and deposited elsewhere. A similar question could be asked of the two other areas studied. The fate of this material from shellfish leases could to a large extent be predicted from computer hydrodynamic models (either 2 or 3 dimensional) with input of various parameters, namely tidal movement, wind speed, wave action, bathymetry and particle settling rates. However, identification of material in sediments stemming from shellfish biodeposits is difficult due to the similarity of source material (seston) composition. Identification of an appropriate tracer would be required. Stable isotopes of carbon, nitrogen and sulphur have been used to trace the fate of material from known sources (e.g. Peterson and Fry, 1987). However, the use of stable isotopes to track biodeposits from shellfish is complicated by the fact that the material expelled by oysters is composed of ambient matter which would also be naturally deposited. Thus, it would be difficult to differentiate between material sourced from shellfish, via their biodeposits, and that which naturally settles from the water column. A study conducted by Riera (1998) showed $\delta^{15}N$ of oysters (*Crassostrea gigas*) tissue reflected their diet with enrichment shown. However, their study did not discuss biodeposits, or $\delta^{15}N$ values of this material. Particulate organic matter and sediment organic matter samples collected from two sites where oysters were present showed similar $\delta^{15}N$ values, with reasonable overlap (Riera, 1998).

6.1 Summary

The study of ecosystem dynamics is a complex and fascinating area of research, with scales of temporal and spatial processes occurring over varying times from seconds to years and distances of mm to kilometres. However, such complete investigations entail mammoth and costly multi-disciplinary investigations. Approaches adopted have been to study components of these processes, in order to provide some insight and understanding of the overall behaviour of system dynamics. Selectivity, clearance and ingestion of particles from the water column has been shown, with pseudofaeces production occurring as a consequence of rejection of particles (selective ingestion) or when particulate matter concentration exceeds a threshold value (Kiørboe and

Møhlenberg, 1981; Shumway et al., 1985; Powell et al., 1992; Barillé and Prou, 1993; Pastoureaud et al., 1996; Hawkins et al., 1997).

The process of filtration, selection and biodeposition in bivalves is complex, and is not simply related to concentration of particulate matter in the water column, but is also related to the quality of the particulate matter, size spectrum of seston material, the capacity of bivalves to adapt to the type of food available, variation in food availability and feeding time (especially for intertidal species which experience periods of restricted inundation), and environmental conditions (e.g. temperature and salinity). Metabolic costs associated with these processes, inclusive of digestion, absorption and transformation of food material, for example, are complex (Willows, 1992), and were beyond the scope of this study. This is an area which is still not fully understood, and where continued investigations are being undertaken in order to gain greater insight and understanding of physiological processes in bivalves (e.g. Bayne, 1998; Grant and Bacher, 1998).

The overall findings from this study indicate that growth rates and productivity of each area are largely influenced by the supply and availability of food. It appears that stocking density and spatial arrangement of leases provide the greater limitation on growth rate in Pitt Water and Pipeclay Lagoon. Summarising each area separately:

Little Swanport is characterised as having the better growth rates and conditions for growth. Food quality, as measured by chlorophyll a and %POM in particular, was higher than the other two sites, and flow rates indicate that a greater quantity of food is reaching a larger proportion of the cultured population. Sources of food may in part be of marine origin, however, high quality food resources occur within the estuary (e.g. Van den Enden, 1994; Brown and McCausland, 1999).

The marine nature of Pipeclay Lagoon suggests that the main source of food supply to the cultured oyster population is of marine origin. However, flow rates and transport of this material over the culture area is insufficient to provide faster growth rates. Stocking density of oysters, and spatial arrangement of the culture area, is most likely responsible for limitation on available food supply to the majority of the population. Sufficient food is available for maintaining metabolic processes, but is insufficient to enable greater storage and hence growth rates.

Similar processes appear to be occurring in Upper Pitt Water, though it seems the greater fraction of food is sourced from within the estuary, rather than being of marine

origin. Sampling during this study was fortunate to coincide with infrequent events of heavy and prolonged rainfalls in the latter part of the year, resulting in flooding of the estuary. The beneficial effects of this were elevated nutrient, chlorophyll a, seston levels and greater increase in oyster dry meat weights, confirming the concerns raised by the oyster farmers with respect to the negative effects of the Craigbourne Dam.

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Appendices

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Appendix 1.1 Time - integrated sample bottle design and evaluation.

Water samples were collected using time-integrated sample bottles. This type of sample bottle was developed during the FRDC project titled "Predictive modelling of carrying capacities of oyster (*Crassostrea gigas*) farming areas in Tasmania" (Crawford et al., 1996) as a modification of the time-integrated water sample bottle designed by Fabris et al. (1982). Further modifications to the design were made at the start of this study to increase sample volume, increase the ease with which to evacuate air from the bottles and to improve sample integrity and operational reliability. Fourteen sample bottles were constructed for this study.

The bottles were constructed from 150 mm diameter PVC pipe (as used for plumbing) and length 453 mm, with end caps secured using PVC solvent cement. Sample capacity was 8 L. A strut, made from a length of 30 mm diameter PVC pipe (cut in half length wise), was placed inside the bottle to reduce bowing of the end caps when the bottle was evacuated. A 50 mm diameter hole was cut near the edge on the top cap in which to insert a silicon stopper (Fig. 1). Silicon stoppers were made using Silastic®. A threaded hole was drilled on the opposing side and a small brass elbow screwed in place (this was the water sample inlet). A short length of Tygon® tubing connected the brass elbow to an air tight turn-screw valve, of the type used for aquarium tank aerators, which was glued to the top.

The sample intake line, which was connected to the valve, consisted of a 40 mm length of 3.2 mm OD (1.5 mm ID) Tygon® tubing, in-line 10 mm ID filter disk (with 500 µm mesh screen insert) and length of 9.0 mm OD tubing (weighted near the end) which hung approximately 1 m below the water surface. The filter disk could be unscrewed to clean, or replace, the screen mesh. The bottles were made air tight by sealing all joins and connections with a sealant glue. Prior to sampling, the bottles were evacuated (to create a vacuum), by disconnecting the sample intake line and connecting a vacuum line (with pressure gauge attached) directly to the valve.

On deployment the bottle was attached to an anchored surface float (Fig. 2) and the intake tube set approximately 1 m below the water surface. The valve at the top of the bottle was opened and the negative pressure (vacuum) in the bottle induced the inflow of water. Flow rate into the bottle was controlled by the length and cross-sectional area

(Fabris et al., 1982) of the Tygon® tubing on the sample intake line. Approximate filling time was 50 minutes, equating to a mean flow rate of 2.7 ml s^{-1}.

Fig. 1. Time-integrated water sample bottles used in this study.

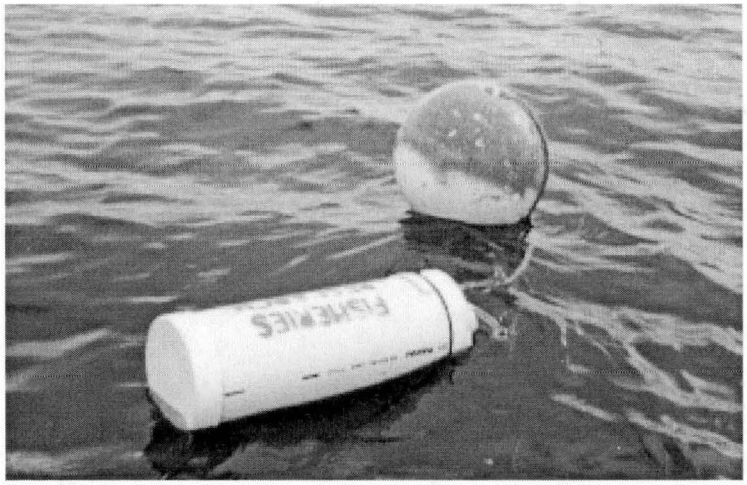

Fig. 2 Sample bottle attached to an anchored 35 cm surface float. Intake line set 1 m sub-surface.

Prior to commencement of sampling in this study, assessment was made of the time-integrated water sample bottles as used in the FRDC study. This was done in order to

improve the design and efficiency of the sample bottles and to evaluate water sample quality to ensure representative sample collection. Preliminary comparison was made between the sample bottles used in the FRDC study and sample bottles with modified intake lines and configuration. The sample bottles used in the FRDC study differed in having a 21 gauge (0.80 x 25 mm) syringe needle (with end removed) sleeved in silicon tubing which passed through a silicon stopper at the top of the sample bottle. A 'press-click' hose clamp sealed the line to maintain vacuum pressure. However, often this system failed and loss of vacuum occurred. The in-line filter and suspended 1 m length of tubing configuration remained unaltered.

Instantaneous grab samples were collected at the start and end of this assessment, with a sub-sample of the initial grab sample replicates (n=3) filtered through a 500 μm mesh screen for comparison. Modification of the intake line included the use of 4.2 mm OD and 3.2 mm OD Tygon® tubing. Results of this assessment are presented in Fig 3.

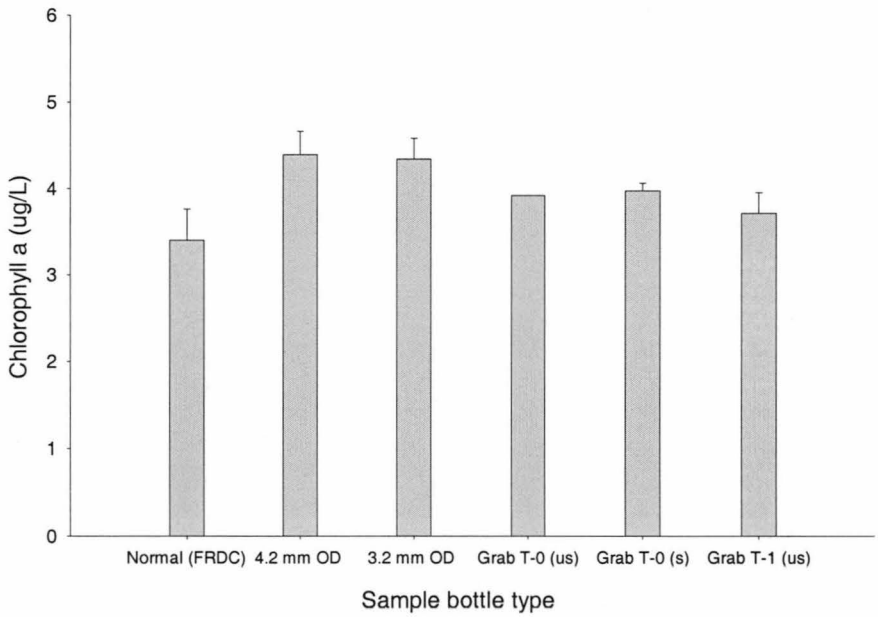

Fig. 3. Mean (± sd) chlorophyll a concentrations in water samples. Normal (FRDC) refers to the sample bottles used in the FRDC study, 4.2 mm OD and 3.2 mm OD refer to modification of the intake line and grab samples at initial and final sample time. us = unscreened, s = screened through 500 μm mesh.

Following this assessment, 3.2 mm OD tubing of 40 mm length was selected for use on the intake line. Whilst values for chlorophyll a were marginally higher using the 4.2 mm OD tubing, the time taken to fill bottles was greater using the narrower (3.2 mm) tube. Hence, with the aim of gaining time-integrated samples, preference was given to the slightly longer fill time achieved using the 3.2 mm OD tubing.

The difference in chlorophyll a concentrations between the screened and unscreened grab samples collected at the start of the trial is unknown. It could be that the greater fraction of phytoplankton was within the less than 200 μm range, and hence per volume filtered the concentration was higher. Further investigation would be required to assess if this was so.

Further assessment was made of the time-integrated sample bottles with respect to total particulate matter and nutrient concentrations. Sampling was conducted in North-West Bay, with replicate samples (n=3) collected across a transect set perpendicular to the incoming flow of water. Time-integrated sample bottles were attached to lines set approximately 20 m apart in 10 m of water depth. Grab samples were collected at the time of deployment of the integrated samplers, and at 20 and 40 mins after deployment from each of the integrated sample bottle locations (n=3). Temperature, salinity and secchi depth were recorded at the start and finish of the trial. Grab samples were screened through 500 μm mesh size to ensure uniformity with the integrated sample bottles.

Temperature and salinity did not change over the 40 minute period (12.2^0 C and 34.2‰, respectively). Secchi depth was 5.2 m at the start and 5.5 m at the finish, with a green colouration noted in the water column. Prevailing weather conditions were cloud cover of $5/8^{ths}$ and west to south-westerly winds of 10-15 knots. The water samples collected were analysed for TPM, POM, PIM, chlorophyll a and nutrients (PO_4-P, NOX-N and SiO_4-Si). The mean values for each of these parameters are shown in Fig. 4.

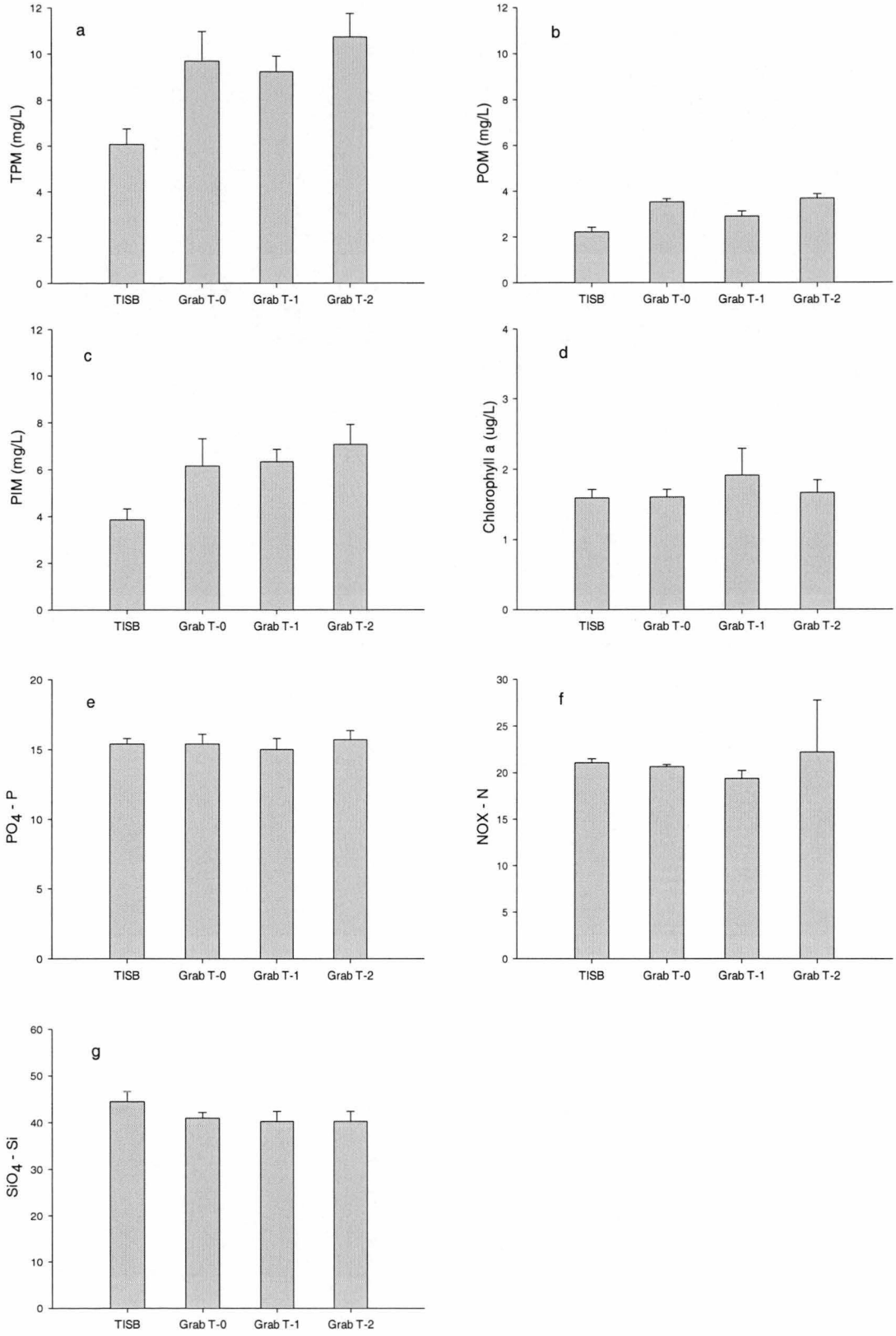

Fig. 4. Comparison of water quality parameters between time-integrated sample bottles (TISB) and grab samples in North-West Bay. Grab samples collected at start (T-0), 20 mins (T-1) and 40 mins (T-2). Mean (± sd) water quality parameters measured were a. TPM (mg/L), b. POM (mg/L), c. PIM (mg/L), d. Chlorophyll a (µg/L), e. PO_4-P (µg/L), f. NOX-N (µg/L) and g. SiO_4-Si (µg/L).

The most notable feature of this assessment was the difference in TPM concentrations and hence POM and PIM values, between the integrated sample bottles and the grab

samples. Chlorophyll a and nutrient concentrations were comparable and indicated that the sample bottles adequately represented the mean values over this time period. The reason for the difference shown in the TPM, POM and PIM values could be due to the different volumes of water filtered. Volumes filtered for TPM determinations were 0.8 L and 0.45 L for the integrated and grab samples, respectively.

It was noted, during assessment of a method for filtering water samples for seston determinations (i.e. TPM, POM and PIM), that greater care had to be taken in the filtration procedure for seston concentrations as compared to chlorophyll a. The volume of sample filtered considerably influenced the value obtained. This was shown in results obtained from filtering seawater at various volumes (Fig. 5). Particle loading on, and retentive capacity of, filters was considered to influence the measurement of seston quantities. Retention of particles smaller than the nominal pore size of filters and increased loading, or clogging of filter pores, can cause overestimation or variable results of particle concentration (Sheldon and Sutcliffe, 1969; Sheldon, 1972; Johnson and Wangersky, 1985). Prolonged filtration times necessary to filter larger sample volumes could lead to loss of integrity of the filter and possibly the pulling through of material retained.

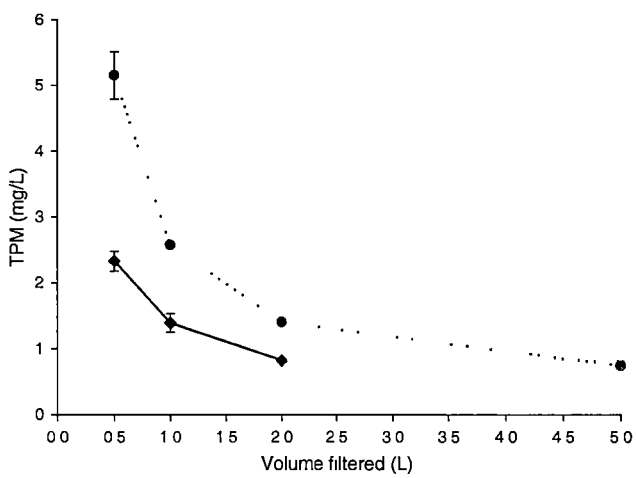

Fig. 5. Mean (± sd) total particulate matter (TPM) mg/L of seawater filtered at various volumes. • undiluted seawater, ♦ 50% diluted seawater. (n=3 for 0.5 and 1.0 L, n=2 - 2.0 L)

Overall, the sample bottles designed proved to be effective in obtaining time-integrated water samples for the parameters measured. They were relatively cheap to construct, easy to deploy and provided adequate volumes for the estimates of the parameters measured. Approximate costing of each sample bottle was $45 ($AUS).

Appendix 1.2

Summary table of Pitt Water Pilot Study (source Crawford et al., 1996). NB: Chlorophyll a values calculated as per the formula of Tett (1987).
Day 1 = 31 October 1994.

Time	TIME	SITE	Rep No.	Temp (C)	Salinity (ppt)	NOX-N	PO4-P	SiO4-Si	CHL a
Day 1	12:50 PM	Shark Pt.	1	15.0	33.8	0.8	10.2	218	0.59
	12:52 PM	Shark Pt.	2	15.0	33.8	0.7	10.0	215	0.54
	12:55 PM	Shark Pt.	3	15 0	33 8	1.0	11 2	205	0.64
Day 2	1:07 PM	Shark Pt.	1	14.8	33.8	1.7	13.0	190	0.93
	1:09 PM	Shark Pt.	2	14.8	33.8	0.9	10 4	170	0.78
	1:12 PM	Shark Pt.	3	14.9	33.8	0.9	10.4	170	0 78
Day 3	1:50 PM	Shark Pt.	1	15.2	33.8	0.4	10.8	251	1 18
	1:52 PM	Shark Pt.	2	15.6	33.9	0.6	10.8	251	1.13
	1:55 PM	Shark Pt.	3	15.4	33.9	0.3	10.4	244	1.03
Day 4	2:54 PM	Shark Pt.	1	15.9	34.0	0.8	11.9	230	1.13
	2:57 PM	Shark Pt.	2	15.8	33.6	1.2	11.2	235	1.23
	2 59 PM	Shark Pt.	3	15.8	33.6	0 8	10.8	254	1 13
Day 5	4.11 PM	Shark Pt.	1	16.2	33.9	0 8	10.8	209	0.88
	4:13 PM	Shark Pt.	2	16.3	33.9	0.8	11.9	228	0.78
	4:15 PM	Shark Pt.	3	16.4	33.9	1 6	11.2	230	0 54
Day 6	5:02 PM	Shark Pt.	1	15.7	33.8	1 9	10.8	220	0 80
	5:04 PM	Shark Pt.	2	15.6	33.8	0 6	10.8	205	0.86
	5:06 PM	Shark Pt.	3	15.6	33.8	1 3	10.8	209	0.92
Week 1	6:06 PM	Shark Pt.	1	14.6	33.8	1.5	11.7	223	0.64
	6:18 PM	Shark Pt.	2	14.4	33.9	0.8	10.0	205	0.69
	6:20 PM	Shark Pt.	3	14.2	33.7	0 9	10.4	200	0.64
Week 2	2:10 PM	Shark Pt.	1	13.8	34.0	0.8	9.2	158	0.43
	2:15 PM	Shark Pt.	2			0 7	10.8	163	0.55
	2:20 PM	Shark Pt.	3	13.6	34.0	2.2	10.4	163	0 06
Week 3	6:12 PM	Shark Pt	1	16.2	34.4	2.6	11.2	185	0.92
	6:14 PM	Shark Pt.	2	15.8	34.5	0.7	11 5	186	0.92
	6:16 PM	Shark Pt.	3	15.8	34.5	1.7	11 9	175	1.23
Week 4	12:18 PM	Shark Pt.	1			0.1	11 5	207	1.02
	12:22 PM	Shark Pt.	2			0.1	10.8	207	0.98
	12:26 PM	Shark Pt.	3						
Day 1	1:10 PM	N. causeway	1	14 3	33.7	0.4	8.8	160	0.44
	1:15 PM	N. causeway	2	14 4	33 6	0.6	9.2	163	0.49
	1:20 PM	N. causeway	3	14.4	33 6	1.1	9.8	171	0.59
Day 2	1:25 PM	N. causeway	1	14.9	33.6	0.9	8.9	126	0.46
	1:28 PM	N. causeway	2	14.9	33.6	1.3	10.6	140	0 59
	1:30 PM	N. causeway	3	14.8	33.7				0.63
Day 3	2:06 PM	N. causeway	1	15.2	33.7	1.5	10.0	209	0.93
	2 08 PM	N. causeway	2	15.2	33.8	0.5	10.0	195	1.03
	2:10 PM	N. causeway	3	15.4	33.7	0.5	10.4	195	1.08
Day 4	3:10 PM	N. causeway	1	15 2	33.2	0.9	8 8	181	1.08
	3.12 PM	N. causeway	2	15.2	33.4	1.8	7.9		0.88
	3:14 PM	N. causeway	3	15.4	33.4	2.2	10.0	193	1.23
Day 5	4:28 PM	N. causeway	1	15.5	33 7	2.2	9.6	184	0.88
	4:30 PM	N causeway	2	15.5	33.7	1.4	10.0	187	0.93
	4:33 PM	N. causeway	3	15.3	33.7	1 1	10.0	156	0.93
Day 6	5.18 PM	N. causeway	1	15.0	33.7	0.9	9.6	186	0.92
	5.20 PM	N. causeway	2	14 5	33.4	0.9	9.6	170	1.16
	5:22 PM	N. causeway	3	14 4	33.4	1.2	9.2	112	0.80

Appendix 1.2 continued

Time	TIME	SITE	Rep No.	Temp (C)	Salinity (ppt)	NOX-N	PO4-P	SiO4-Si	CHL a
Week 1	6:35 PM	N. causeway	1	14.1	33.8	1.4	12.6	209	0.61
	6:37 PM	N causeway	2	14.0	33.6				0.74
	6:40 PM	N causeway	3						
Week 2	3:10 PM	N. causeway	1	13.1	33.8	0.9	9.2	140	0.61
	3:12 PM	N causeway	2	13.6	33.9	1.1	8.5	131	0.49
	3:14 PM	N causeway	3	13.8	34.0	1.4	9.6	121	0.55
Week 3	6:25 PM	N. causeway	1	15.2	34.2	1.1	11.2	157	1.59
	6:27 PM	N causeway	2	15.1	34.3	0 4	10.8	156	1.53
	6:29 PM	N. causeway	3	14.9	34.2	2.3	10.8	144	1.35
Week 4	11:56 AM	N. causeway	1			4.6	10.0	164	1.23
	11:58 AM	N. causeway	2			0.4	10.0	164	1 13
	12:01 PM	N. causeway	3						

Appendix 1.3.1 Pitt Water

Date	Site name	Site No.	Rep no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
28/Feb/95	Marine	1	A	18.5	33.6	2.8	7 26	2.97	4.29	40.91	59.09	1.286	0.8	8.7	78	0.6	6.0
28/Feb/95	Marine	1	B	18.5	33.6	2 8	5.34	2 10	3.24	39.33	60.67	1.562	1.3	10.0	115	0.6	6 0
28/Feb/95	Lewisham	2	A	16 2	34.7	3.8	7.74	2 37	5 37	30.62	69.38	1 011	1.9	12.6	242	0 6	6 0
28/Feb/95	Lewisham	2	B	16 2	34.7	3.8	8.48	2.51	5.97	29.60	70.40	1.378	2.1	13.0	258	0.6	6.0
28/Feb/95	Woody Is	3	A	16.2	34.9	1 8	10.11	2 59	7 52	25.62	74 38	1 470	2 5	14 4	295	0 6	6 0
28/Feb/95	Woody Is	3	B	16 2	34.9	1 8	11.29	2.87	8.42	25.42	74 58	1.011	2 1	14 8	322	0.6	6 0
28/Feb/95	N Causeway	4	A	15.9	34 9	1.8	12.52	3 74	8 78	29 87	70 13	1 011	2 9	14 4	256	0 6	6.0
28/Feb/95	N Causeway	4	B	15.9	34.9	1.8	13.60	3.98	9 62	29.26	70.74	1.286	2 9	14.1	258	0 6	6 0
28/Feb/95	Barilla Bay	5	A	18.8	36 4	1.2	12.38	4 60	7 78	37 16	62 84	0.735	1 0	9.8	217	0.6	6 0
28/Feb/95	Barilla Bay	5	B	18.8	36 4	1.2	11.22	4.32	6.90	38 50	61.50	0.827	1.3	9.6	207	0 6	6 0
28/Feb/95	Shark Pt	6	A	16.5	36 0	1.5	16.16	4 66	11 50	28 84	71.16	1.654	3.1	15 6	315	0.6	6 0
28/Feb/95	Shark Pt.	6	B	16.5	36 0	1.5	18.62	5 22	13.40	28 03	71 97	1.746	3.5	16 3	315	0 6	6 0
28/Feb/95	Top end	7	A	18.7	36 5	0.6	56.67	11.17	45 50	19.71	80.29	2.389	3.1	20.8	371	0.6	6 0
28/Feb/95	Top end	7	B	18 7	36.5	0.6	60.20	12.65	47 55	21 01	78 99	2 021	3.7	17.7	457	0.6	6 0
12/Apr/95	Marine	1	A	12.1	34.6	2 5										65 6	66 6
12/Apr/95	Marine	1	B	12 1	34 6	2.5	11.03	3 93	7 10	35 62	64.38	1.194	3.7	13.8	82	65.6	66 6
12/Apr/95	Lewisham	2	A	10.7	33 6	3.3	7.73	2 70	5 03	34.91	65 09	0.919	11.7	10.0	216	65.6	66.6
12/Apr/95	Lewisham	2	B	10.7	33 6	3 3	7.78	2 95	4 83	37 90	62 10	0 919	11 5	10.0	221	65 6	66.6
12/Apr/95	Woody Is.	3	A	11 2	35 5	1.6	9.07	3 08	5 98	34 01	65.99	1 286	1 7	11 5	184	65 6	66 6
12/Apr/95	Woody Is	3	B	11.2	35 5	1 6	10.18	3 23	6 95	31 75	68.25	1 194	0.6	10.0	196	65.6	66.6
12/Apr/95	N Causeway	4	A	11.1	35 5	2 2	9.08	2 90	6 18	31.93	68 07	1 470	1 3	10.2	171	65.6	66 6
12/Apr/95	N. Causeway	4	B	11 1	35 5	2.2	9.87	3 05	6 82	30 91	69.09	1 378	1.2	12 7	171	65.6	66.6
12/Apr/95	Barilla Bay	5	A	10 7	35.7	1.2	9.27	2 83	6 43	30.58	69.42	0 551	1 3	8.5	152	65.6	66.6
12/Apr/95	Barilla Bay	5	B	10.7	35.7	1.2	12.53	3 43	9.10	27.39	72 61	0 551	1 9	10.0	152	65.6	66 6
12/Apr/95	Shark Pt	6	A	10 6	35.5	1.6	12.90	3.90	9.00	30.23	69.77	1 103	1 5	10 0	180	65.6	66 6
12/Apr/95	Shark Pt.	6	B	10.6	35 5	1.6	12.84	4 04	8 80	31.46	68.54	1 378	2 4	9.6	182	65.6	66 6
12/Apr/95	Top end	7	A	10.8	34.8	1.4	12.72	3 66	9 06	28 77	71 23	1 011	1 5	10 2	223	65.6	66 6
12/Apr/95	Top end	7	B	10.8	34 8	1.4	12.58	3.22	9.36	25.60	74.40	0 919	2.2	10 2	198	65.6	66.6
24/May/95	Marine	1	A	11.1	34 7	6.8	9.51	3 07	6.44	32.28	67.72	0 459	2.5	11.3	120	0 0	2.2
24/May/95	Marine	1	B	11.1	34 7	6.8	5.47	2 14	3 33	39 16	60 84	1.011	2 6	10 8	148	0 0	2.2
24/May/95	Lewisham	2	A	10.2	34.8	6.5	6.54	2 52	4.02	38 53	61 47	0 092	3.3	10 8	138	0 0	2.2
24/May/95	Lewisham	2	B	10.2	34.8	6.5	6 50	2.32	4.18	35.69	64 31	0 459	3 5	11.5	140	0.0	2 2
24/May/95	Woody Is	3	A	10.0	35.3	1.5	18.22	4.22	14.00	23.16	76.84	0 735	0.5	11.2	175	0.0	2.2
24/May/95	Woody Is.	3	B	10 0	35.3	1.5	7.42	2.40	5.02	32.35	67 65	0.551	0.5	9 6	200	0.0	2 2
24/May/95	N Causeway	4	A	10.2	35 2	1 8	8.80	2 52	6 28	28.64	71 36	1.103	3.3	10 4	175	0 0	2.2
24/May/95	N. Causeway	4	B	10.2	35 2	1.8	8.00	2 72	5 28	34.00	66 00	0.643	1.0	9.6	178	0.0	2.2
24/May/95	Barilla Bay	5	A	8.2	35 6	1.2	7 46	2 44	5 02	32 71	67.29	0.551	1.5	8 7	130	0.0	2.2
24/May/95	Barilla Bay	5	B	8.2	35 6	1.2	8.18	3.02	5 16	36.92	63.08	-0 184	1.4	8 5	132	0.0	2.2
24/May/95	Shark Pt	6	A	9.2	35 4	1.8	9.12	2 44	6 68	26 75	73.25	0.827	4.7	10 0	175	0 0	2 2
24/May/95	Shark Pt.	6	B	9.2	35 4	1 8	9.46	2 66	6 80	28 12	71 88	0 459	2 5	10 0	172	0 0	2 2

Date	Site name	Site No.	Rep no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
24/May/95	Top end	7	A	8.6	35.4	1.4	9 54	2.54	7.00	26 62	73.38	1.746	0.8	9 6	182	0.0	2 2
24/May/95	Top end	7	B	8.6	35.4	1 4	10 42	2.46	7.96	23.61	76.39	-0.368	0.6	9 6	190	0.0	2.2
22/Jun/95	Marine	1	A	8.7	34 3	3 2	14 82	4.28	10.54	28.88	71.12	2.297	2.0	9 0	18	6.6	8 6
22/Jun/95	Marine	1	B	8 7	34 3	3.2	7 70	2.70	5.00	35 06	64 94	2.757	0.0	9 0	30	6.6	8.6
22/Jun/95	Lewisham	2	A	7.0	34.4	4.5	6.42	2.28	4 14	35 51	64.49	-0.368	4 0	8 5	54	6 6	8.6
22/Jun/95	Lewisham	2	B	7.0	34.4	4.5	6 44	2 22	4 22	34.47	65.53	0 919	1.9	9.0	124	6.6	8 6
22/Jun/95	Woody Is	3	A	6.2	34 9	1.5	7 02	2.22	4 80	31 62	68 38	0.092	1 1	8 0	82	6 6	8 6
22/Jun/95	Woody Is.	3	B	6 2	34 9	1.5	6.32	1.98	4.34	31 33	68 67	0 459	1 1	8 0	136	6 6	8 6
22/Jun/95	N Causeway	4	A	6 4	34 7	3.2	6.44	2 16	4.28	33 54	66 46	0 643	1 6	8 0	86	6 6	8 6
22/Jun/95	N Causeway	4	B	6 4	34.7	3.2	6.72	2 30	4 42	34 23	65 77	0 276	3.1	8 0	80	6.6	8.6
22/Jun/95	Barilla Bay	5	A	4 4	35 3	1.2	7.40	2 38	5.02	32 16	67 84	0 551	1 5	8 0	83	6 6	8 6
22/Jun/95	Barilla Bay	5	B	4 4	35 3	1.2	7.32	2 30	5.02	31 42	68 58	0 459	2 1	8 0	82	6 6	8.6
22/Jun/95	Shark Pt	6	A	5 8	34 9	2 6	8.10	2 32	5.78	28 64	71 36	0 827	1 5	8 0	96	6 6	8.6
22/Jun/95	Shark Pt	6	B	5.8	34 9	2.6	7 04	1 96	5 08	27.84	72 16	0 919	3.5	7 0	114	6 6	8 6
22/Jun/95	Top end	7	A	5 0	35 5	1.8	9 80	2.66	7 14	27 14	72.86	0 643	2 6	7.0	118	6.6	8 6
21/Jun/91	Top end	7	B	5 0	35.5	1 8	9 26	2 28	6 98	24 62	75.38	1 011	1.5	7.0	116	6 6	8 6
22/Aug/95	Marine	1	A	8 9	33.5	3 5	6 12	2 10	4 02	34.33	65 67	1 838	0 6	8 1	30	15.8	15 8
22/Aug/95	Marine	1	B	8.9	33.5	3.5										15.8	15.8
22/Aug/95	Lewisham	2	A	9.7	32 9	5.0	6 58	2.17	4 42	32.91	67.09	1.303	2.2	7 3	44	15 8	15 8
22/Aug/95	Lewisham	2	B	9.7	32 9	5.0	5 35	1.87	3 48	34 89	65.11	1 403	0.7	6 9	44	15.8	15 8
22/Aug/95	Woody Is.	3	A	10.1	32 3	1.8	6 74	2 32	4.42	34.42	65.58	0.401	0.6	6 2	70	15 8	15.8
22/Aug/95	Woody Is.	3	B	10.1	32.3	1 8	6.30	2 20	4 10	34.92	65.08	1 002	0.4	5 8	85	15.8	15.8
22/Aug/95	N Causeway	4	A	9 4	31.9	2 2	6 38	2.20	4 18	34.48	65.52	1.203	1 7	6 2	78	15.8	15.8
22/Aug/95	N Causeway	4	B	9.4	31.9	2.2	7.00	2.36	4.64	33 71	66.29	1.203	1 3	5 8	85	15.8	15.8
22/Aug/95	Barilla Bay	5	A	10.3	31 6	1.6	10.14	3.30	6.84	32.54	67.46	1.504	0 7	4.6	78	15.8	15.8
22/Aug/95	Barilla Bay	5	B	10.3	31 6	1.6	8 74	2.84	5.90	32.49	67 51	1.604	0.0	5.0	76	15.8	15 8
22/Aug/95	Shark Pt.	6	A	9 9	31 9	2.5	7.14	2.26	4.88	31 65	68 35	1.103	0 4	6 2	109	15 8	15 8
22/Aug/95	Shark Pt	6	B	9.9	31 9	2.5	8 18	2.60	5 58	31.78	68.22	1 103	0 9	5 4	104	15.8	15 8
22/Aug/95	Top end	7	A	9 9	31.5	1.4	10.92	3 10	7.82	28 39	71 61	2 005	0.4	6 2	137	15 8	15 8
22/Aug/95	Top end	7	B	9 9	31 5	1.4	10.70	2.72	7 98	25 42	74 58	2.205	0.8	6.7	123	15.8	15.8
26/Sep/95	Marine	1	A	10.8	34 0	2.5	11.26	3.30	7 96	29.31	70.69	1.103	2.0	10.8	58	2 2	2 2
26/Sep/95	Marine	1	B	10.8	34 0	2 5	8.94	2.88	6 06	32 21	67.79	1.403	1.7	10 0	70	2.2	2.2
26/Sep/95	Lewisham	2	A	11.0	34 1	3.5	7 66	2.66	5 00	34.73	65.27	0.702	2 8	7.7	60	2.2	2 2
26/Sep/95	Lewisham	2	B	11.0	34 1	3 5	7 92	2 60	5 32	32.83	67.17	0.902	1 3	7.7	65	2.2	2.2
26/Sep/95	Woody Is	3	A	10.5	33 6	1.5	6 98	2.18	4 80	31.23	68.77	0.200	0.7	6 9	88	2 2	2 2
26/Sep/95	Woody Is.	3	B	10.5	33 6	1 5	6 64	1.96	4 68	29.52	70.48	0.401	3 9	8 5	100	2.2	2.2
26/Sep/95	N Causeway	4	A	10.5	33.6	1 9	8.50	2 40	6 10	28.24	71.76	0 301	1 3	6.9	88	2.2	2 2
26/Sep/95	N. Causeway	4	B	10.5	33.6	1 9	9 26	2 46	6 80	26.57	73.43	0.501	0 6	7 3	100	2.2	2.2
26/Sep/95	Barilla Bay	5	A	12 5	33.6	1 5	10 70	3 64	7.06	34.02	65.98	-0 100	0 6	8.8	123	2.2	2.2
26/Sep/95	Barilla Bay	5	B	12 5	33.6	1 5	8.50	2 26	6.24	26 59	73 41	0 000	0.7	7.7	115	2 2	2.2
26/Sep/95	Shark Pt.	6	A	11.2	33 6	2.2	7 08	1.84	5.24	25.99	74.01	0.601	0.6	7.7	120	2.2	2 2
26/Sep/95	Shark Pt.	6	B	11.2	33 6	2.2	7.94	2.12	5.82	26 70	73 30	0.401	0.7	7 7	123	2 2	2.2
26/Sep/95	Top end	7	A	11.8	33 6	1 4	11 56	2 46	9 10	21 28	78.72	0.301	0.4	7.4	140	2 2	2 2

Date	Site name	Site No	Rep. no	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
26/Sep/95	Top end	7	B	11 8	33 6	1 4	10.66	2.14	8.52	20 08	79 92	0 401	1.5	8 5	138	2.2	2.2
24/Oct/95	Marine	1	A	14.0	34.3	3.5	7.90	2.50	5.40	31.65	68 35	0.601	0.6	8 8	31	4.8	22 0
24/Oct/95	Marine	1	B	14 0	34.3	3.5	7.74	2 14	5 60	27.65	72 35	0.501	0.2	9.2	31	4.8	22.0
24/Oct/95	Lewisham	2	A	14 5	34.4	3.2	7.92	2 36	5 56	29 80	70.20	0 601	1.0	8.8	52	4 8	22.0
24/Oct/95	Lewisham	2	B	14.5	34 4	3 2	7.70	2 30	5.40	29 87	70 13	1.002	1.4	8 4	60	4 8	22 0
24/Oct/95	Woody Is	3	A	15 0	34 6	1.5	9 88	2 34	7 54	23.68	76.32	0 902	1.0	8.8	105	4 8	22 0
24/Oct/95	Woody Is	3	B	15 0	34 6	1.5	9.06	2 30	6.76	25.39	74 61	1.303	0.4	8.1	132	4.8	22 0
24/Oct/95	N. Causeway	4	A	15.1	34.6	2 4	8 02	2.12	5.90	26 43	73.57	0.702	0 9	8 1	110	4.8	22.0
24/Oct/95	N Causeway	4	B	15 1	34.6	2.4	8 28	2.20	6 08	26.57	73.43	0 601	0.2	8.1	117	4.8	22.0
24/Oct/95	Barilla Bay	5	A	17 0	34.9	1 1	10.22	3 28	6.94	32.09	67 91	0 200	1.4	8 4	137	4 8	22 0
24/Oct/95	Barilla Bay	5	B	17 0	34.9	1 1	16 00	6 90	9 10	43.13	56 87	0.601	0 6	8 1	137	4 8	22 0
24/Oct/95	Shark Pt.	6	A	15 7	34 6	1.7	9.66	2.20	7.46	22 77	77.23	0.802	0 8	9.2	156	4 8	22 0
24/Oct/95	Shark Pt.	6	B	15.7	34 6	1 7	9.92	2 28	7.64	22 98	77 02	0 601	0.5	8 8	159	4.8	22 0
24/Oct/95	Top end	7	A	16.6	34 8	0 9	16.46	2 80	13.66	17.01	82 99	0 802	0.8	9 6	190	4.8	22 0
24/Oct/95	Top end	7	B	16.6	34 8	0 9	19.20	3.40	15 80	17 71	82.29	0.902	1.2	9 6	190	4 8	22.0
22/Nov/95	Marine	1	A	14.5	34 3	5.5										3 4	10.4
22/Nov/95	Marine	1	B	14 5	34.3	5 5	8.22	2 54	5.68	30 90	69.10	1.303	0.9	8.5	59	3 4	10.4
22/Nov/95	Lewisham	2	A	15 8	34.3	2 0	8 60	2 34	6.26	27.21	72 79	1 103	2.0	8.1	149	3 4	10.4
22/Nov/95	Lewisham	2	B	15.8	34 3	2.0	10 12	2 62	7.50	25.89	74 11	1 303	0.8	8.1	159	3.4	10.4
22/Nov/95	Woody Is.	3	A	15 5	34 4	1.2	14.10	3 80	10.30	26 95	73 05	1 203	0 9	9.2	198	3.4	10 4
22/Nov/95	Woody Is	3	B	15 5	34.4	1 2	14.88	3 80	11.08	25 54	74 46	1.403	1.1	8 8	220	3.4	10 4
22/Nov/95	N. Causeway	4	A	15 4	34 4	1 4	10.98	2.68	8.30	24 41	75.59	1.504	0 8	8 1	175	3 4	10 4
22/Nov/95	N. Causeway	4	B	15.4	34 4	1.4	12 24	2.78	9.46	22.71	77 29	1 604	0.3	8.5	200	3.4	10.4
22/Nov/95	Barilla Bay	5	A	17.8	34 6	0 8	34.70	7.48	27 23	21 54	78 46	0 802	2.0	8.8	222	3.4	10.4
22/Nov/95	Barilla Bay	5	B	17 8	34.6	0 8	97.50	15.30	82 20	15 69	84 31	1 851	2.3	9.6	243	3 4	10.4
22/Nov/95	Shark Pt	6	A	16.7	34.6	1 3	13 45	3 52	9 93	26 21	73.79	0.802	1.3	8 8	286	3 4	10.4
22/Nov/95	Shark Pt	6	B	16 7	34.6	1.3	12 82	3 06	9 76	23.87	76 13	1 103	1 0	9 2	280	3 4	10.4
22/Nov/95	Top end	7	A	16 9	34.6	0.4	259 60	25.55	234 05	9.84	90.16	1 804	1 6	10 0	334	3 4	10.4
22/Nov/95	Top end	7	B	16 9	34.6	0.4										3.4	10 4
29/Dec/95	Marine	1	A	20.0	34.5	3.00										3 2	92.0
29/Dec/95	Marine	1	B	20.0	34 5	3.00	15.90	4.54	11.36	28.55	71.45	0.902	0 5	9 2	72	3 2	92.0
29/Dec/95	Lewisham	2	A	20 0	32 5	2.50	7.96	2.92	5 04	36.68	63.32	1 403	0 5	4 6	308	3 2	92.0
29/Dec/95	Lewisham	2	B	20 0	32 5	2.50	8 52	2 92	5 60	34.27	65.73	1 604	1 0	4 6	308	3 2	92.0
29/Dec/95	Woody Is	3	A	19.0	30.0	1.50	14 10	3 72	10.38	26.38	73 62	1 604	2.6	6.3	476	3.2	92 0
29/Dec/95	Woody Is	3	B	19 0	30.0	1.50	13.24	3 70	9 54	27 95	72.05	1.804	0.6	7 1	536	3.2	92 0
29/Dec/95	N. Causeway	4	A	19 0	30 0	1.50	10 86	3 42	7 44	31.49	68.51	2 105	0 8	5 0	500	3 2	92.0
29/Dec/95	N Causeway	4	B	19.0	30.0	1.50	10.86	3.54	7.32	32 60	67.40	2 005	0 8	5 0	500	3 2	92.0
29/Dec/95	Barilla Bay	5	A	20 0	30 0	1.00	25 12	5 50	19.62	21 89	78.11	1.002	1.5	5 4	460	3.2	92.0
29/Dec/95	Barilla Bay	5	B	20.0	30 0	1.00	9.40	3.72	5.68	39.57	60 43	0.902	1.5	4.2	456	3.2	92 0
29/Dec/95	Shark Pt.	6	A	18.0	29.5	1 50	12.38	3 84	8 54	31 02	68 98	1 704	1 1	5.8	564	3.2	92.0
29/Dec/95	Shark Pt	6	B	18.0	29 5	1 50	12.28	3.84	8.44	31.27	68 73	1 504	1 1	5 4	560	3.2	92 0
29/Dec/95	Top end	7	A	18 9	28 0	0.95	21.94	5 50	16 44	25.07	74.93	1.002	0 3	5 8	720	3.2	92.0
29/Dec/95	Top end	7	B	18 9	28.0	0 95	19.67	4 97	14.69	25 29	74.71	1.871	1 3	5.8	720	3 2	92.0

Date	Site name	Site No.	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
19/Jan/96	Marine	1	A	18.4	34.2	4.5										0.6	1 4
19/Jan/96	Marine	1	B	18.4	34.2	4.5	8.88	2.66	6 22	29.95	70 05	0 802	1.4	6.8	104	0 6	1 4
19/Jan/96	Lewisham	2	A	21.4	31 4	1.6	13 72	3.30	10.42	24 05	75.95	1.704	1.8	8 8	258	0.6	1.4
19/Jan/96	Lewisham	2	B	21 4	31.4	1 6	13.02	3.38	9 64	25 96	74 04	1 604	0 8	8 4	258	0 6	1.4
19/Jan/96	Woody Is.	3	A	21 2	30 6	1 1	115 95	15 55	100 40	13 41	86 59	3.475	1.1	22.4	302	0 6	1 4
19/Jan/96	Woody Is	3	B	21 2	30 6	1.1	32.86	5.57	27.29	16 96	83.04	2 005	0 4	12.8	320	0 6	1.4
19/Jan/96	N Causeway	4	A	21 0	30 4	1.6	11.90	3.06	8 84	25.71	74.29	2 105	0 7	8 4	296	0 6	1.4
19/Jan/96	N Causeway	4	B	21 0	30 4	1.6	11.98	3.12	8.86	26 04	73.96	1.804	0 9	8.0	302	0 6	1 4
19/Jan/96	Barilla Bay	5	A	23.8	30.1	1.2	64 40	8.40	56.00	13 04	86 96	2 005	0.7	16.0	311	0 6	1.4
19/Jan/96	Barilla Bay	5	B	23.8	30 1	1 2	16 60	3.28	13.32	19.76	80 24	0 802	1.1	6 0	298	0.6	1.4
19/Jan/96	Shark Pt.	6	A	22 3	29.6	1.3	11 96	2 62	9 34	21.91	78.09	1.704	0.5	8.4	347	0 6	1 4
19/Jan/96	Shark Pt	6	B	22.3	29.6	1 3	10 14	2.22	7 92	21.89	78.11	1 704	0.2	8 8	344	0 6	1.4
19/Jan/96	Top end	7	A	22 1	28.9	0 7	25.71	4 16	21.55	16 19	83 81	2 272	0.2	9.6	391	0.6	1.4
19/Jan/96	Top end	7	B	22 1	28 9	0.7	21 93	4.00	17 93	18 24	81.76	1.737	0.7	6 5	396	0 6	1 4
16/Feb/96	Marine	1	A	17 3	33.1	4 0	14.72	5 56	9 16	37.77	62 23	0 802	1.2	8 5	192	47.2	51.4
16/Feb/96	Marine	1	B	17 3	33.1	4.0	8 18	2.80	5.38	34 23	65 77	0 802	1 2	7 7	188	47 2	51.4
16/Feb/96	Lewisham	2	A	18.3	27 9	2 0	9 32	2 46	6 86	26.39	73 61	2.305	4 0	6.2	846	47.2	51.4
16/Feb/96	Lewisham	2	B	18 3	27.9	2.0	10 48	2.64	7.84	25.19	74.81	2 305	2.7	6 5	862	47.2	51 4
16/Feb/96	Woody Is	3	A	18.4	25 4	1.0	180.29	26.76	153 53	14 85	85.15	6 950	7.3	14 3	1077	47 2	51.4
16/Feb/96	Woody Is.	3	B	18 4	25.4	1.0	38.91	7 51	31 40	19.31	80 69	2.406	5 2	11 4	1015	47.2	51.4
16/Feb/96	N. Causeway	4	A	18 6	24 8	1.4	11 10	2.63	8 47	23 65	76 35	2 138	5.6	6.2	1108	47.2	51 4
16/Feb/96	N. Causeway	4	B	18.6	24 8	1 4	10.68	2 90	7 77	27 17	72 83	2 005	5 6	6.2	1000	47.2	51 4
16/Feb/96	Barilla Bay	5	A	18.9	24.7	1 2	124 85	14 35	110 50	11.49	88.51	2.438	6 0	5.4	1139	47.2	51.4
16/Feb/96	Barilla Bay	5	B	18.9	24 7	1 2	37 33	8 33	29 00	22 32	77 68	1.337	6.3	9.6	1108	47 2	51 4
16/Feb/96	Shark Pt	6	A	19 2	23.2	1.1	10.88	3.02	7 85	27.82	72 18	2 706	2.7	5 0	1123	47.2	51 4
16/Feb/96	Shark Pt	6	B	19.2	23 2	1 1	10 30	3.26	7.04	31.65	68.35	2.907	3 1	5.0	1123	47.2	51 4
16/Feb/96	Top end	7	A	19 5	20 8	0 9	15.40	4.02	11.38	26 14	73 86	2 940	1.0	5.8	1308	47 2	51.4
16/Feb/96	Top end	7	B	19.5	20.8	0 9	15 43	3 60	11 83	23.34	76 66	3 475	1 3	5.4	1277	47 2	51.4
16/Mar/96	Marine	1	A	16 4	34 1	4 0	14.72	4.42	10.30	30 03	69.97	1.504	3 4	8 5	48	12.0	12 0
16/Mar/96	Marine	1	B	16 4	34.1	4 0	7 80	2 64	5 16	33 85	66 15	1 504	1.4	7 7	48	12 0	12.0
16/Mar/96	Lewisham	2	A	16.8	32 2	2 5	8.76	2 60	6.16	29.68	70.32	1.403	4.5	10.0	164	12.0	12.0
16/Mar/96	Lewisham	2	B	16 8	32.2	2.5	9.04	2.66	6 38	29 42	70.58	1.103	4 0	9.8	150	12 0	12 0
16/Mar/96	Woody Is	3	A	17.1	31.6	1 3	120 67	16.20	104.47	13.43	86.57	2 005	1 2	24.2	240	12 0	12.0
16/Mar/96	Woody Is.	3	B	17 1	31 6	1.3	10.52	2.60	7.92	24 71	75 29	1.704	3 0	11.2	234	12.0	12 0
16/Mar/96	N. Causeway	4	A	17 2	31.6	1.6	9.84	2.72	7.12	27 64	72.36	1 303	2 6	11.5	252	12 0	12 0
16/Mar/96	N Causeway	4	B	17.2	31 6	1 6	10.20	2 76	7.44	27.06	72.94	1 303	3.3	11.5	224	12 0	12 0
16/Mar/96	Barilla Bay	5	A	17 1	31.2	1.4	9.20	2.54	6 66	27 61	72.39	0.802	2.2	10.8	250	12 0	12.0
16/Mar/96	Barilla Bay	5	B	17.1	31.2	1.4	9.44	2.62	6 82	27.75	72.25	0 802	1.7	11.2	248	12.0	12 0
16/Mar/96	Shark Pt	6	A	16 8	30.8	1.5	10.94	2 78	8 16	25 41	74 59	1 704	2 3	11.9	280	12 0	12.0
16/Mar/96	Shark Pt.	6	B	16.8	30 8	1 5	9 24	2 40	6.84	25 97	74.03					12 0	12.0
16/Mar/96	Top end	7	A	16 7	30.6	0.8	19.04	3 46	15.58	18.17	81.83	1.203	6.9	13 5	333	12.0	12 0
16/Mar/96	Top end	7	B	16.7	30 6	0.8	18 80	3 40	15 40	18.09	81 91	1.403	3.6	13 6	338	12.0	12.0

Appendix 1.3.2 Pipeclay Lagoon

Date	Site name	Site No.	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day R'fall	12 day R'fall
1/Mar/95	Marine	1	A	17 5	34.1	6.5	6 09	2 58	3.51	42 30	57 70	0 190	0.8	6.3	24	1.4	9.2
1/Mar/95	Marine	1	B	17 5	34 1	6.5										1.4	9.2
1/Mar/95	Cottage	2	A	18.5	34 6	2.0	6.25	2.53	3 72	40.40	59.60	0.381	2 3	9 6	81	1.4	9.2
1/Mar/95	Cottage	2	B	18.5	34 6	2 0										1.4	9.2
1/Mar/95	Bens Gutter	3	A	19 3	34.7	1.5	14 54	5.14	9.40	35 35	64.65	0.856	2 3	9 6	76	1.4	9.2
1/Mar/95	Bens Gutter	3	B	19 3	34 7	1 5	7 90	3 94	3.96	49.87	50.13	0 095	2.1	9 2	91	1.4	9 2
1/Mar/95	Honeywood	4	A	18.0	34 1	2 5	7 64	2.94	4 70	38 46	61 54	0 571	2.0	8 8	95	1.4	9 2
1/Mar/95	Honeywood	4	B	18.0	34.1	2.5	7.25	3 27	3.98	45.06	54.94	0.571	1.9	8.5	86	1 4	9.2
1/Mar/95	Nemo	5	A	18 0	34 0	1.5	17.43	6 75	10.68	38 74	61.26	0 761	2.1	8.8	112	1.4	9.2
1/Mar/95	Nemo	5	B	18.0	34 0	1.5	10 24	4.34	5.90	42.38	57.62	0.476	2.1	9 2	152	1.4	9.2
26/Apr/95	Marine	1	A	12.9	34.5	1.5	16.44	4.73	11 71	28 76	71.24	1.142	3 8	14 4	150	25.2	26.6
26/Apr/95	Marine	1	B	12 9	34 5	1 5	12 47	3 77	8.70	30 24	69 76	1.047	3 5	16.5	196	25.2	26 6
26/Apr/95	Cottage	2	A	11 3	34.4	4 2	5 77	2.07	3.70	35.89	64.11	0.666	2 3	9 2	98	25 2	26.6
26/Apr/95	Cottage	2	B	11 3	34 4	4 2	6 83	2.27	4.56	33 26	66 74	0 476	1 3	10 4	75	25.2	26.6
26/Apr/95	Bens Gutter	3	A	11 2	34.4	1.2	13 11	3.69	9.43	28 10	71 90	0 856	1.2	12 9	73	25 2	26 6
26/Apr/95	Bens Gutter	3	B	11 2	34.4	1.2	27.82	7 80	20.02	28 04	71.96	1 142	2.7	17.1	118	25 2	26 6
26/Apr/95	Honeywood	4	A	11 4	34 3	2.2										25.2	26 6
26/Apr/95	Honeywood	4	B	11 4	34 3	2.2	131.91	25 22	106 70	19.12	80 88	5.138	3.3	20 8	98	25 2	26 6
26/Apr/95	Nemo	5	A	11.3	34 4	2.0	9.65	3 50	6.15	36.27	63 73	0.856	3 5	8.1	64	25.2	26 6
26/Apr/95	Nemo	5	B	11.3	34 4	2.0	10 90	3 68	7.22	33.72	66.28	0.476	3.6	8 8	66	25.2	26 6
23/May/95	Marine	1	A	11.7	34 2	4 4	9 31	2 61	6.70	28 07	71.93	1.332	3.5	10 6	112	0 0	1.2
23/May/95	Marine	1	B	11.7	34 2	4 4	5 09	1.81	3.27	35 67	64.33	0 856	1 3	10 0	90	0.0	1.2
23/May/95	Cottage	2	A	10.1	34.8	3.2	6 24	2.34	3.90	37.50	62 50	0.571	5 9	11.3	93	0.0	1 2
23/May/95	Cottage	2	B	10.1	34.8	3 2	7.34	2.64	4.70	35 97	64.03	0 571	4.0	10 0	93	0 0	1.2
23/May/95	Bens Gutter	3	A	10 0	34 7	1.4	8 10	2.44	5 66	30.12	69 88	0.476	4.8	11 5	90	0 0	1 2
23/May/95	Bens Gutter	3	B	10 0	34 7	1.4	8 06	2.62	5 44	32 51	67 49	0.476	5.0	11.5	90	0.0	1 2
23/May/95	Honeywood	4	A	10.5	34.8	2.5	8 66	2 76	5 90	31 87	68 13	0 476	6.0	10 8	88	0.0	1.2
23/May/95	Honeywood	4	B	10.5	34.8	2 5	8.43	2 55	5.88	30 27	69.73	0.571	6.0	10 8	93	0.0	1.2
23/May/95	Nemo	5	A	10 9	34.7	1.6	6.98	2 08	4 90	29.80	70.20	0.571	6.3	10 4	93	0.0	1.2
23/May/95	Nemo	5	B	10.9	34.7	1.6	10.26	2 68	7.58	26 12	73.88	0 571	5 9	10 6	98	0 0	1.2
26/Jun/95	Marine	1	A	9.3	34.3	3 2	12 70	4.04	8.66	31.81	68.19	3.140	16.0	12.0	61	12.4	23 0
26/Jun/95	Marine	1	B	9.3	34.3	3.2	7 02	2.86	4 16	40.74	59 26	3 521	14.0	13.0	64	12 4	23.0
26/Jun/95	Cottage	2	A	9 1	34 3	1.8	38.72	9 88	28.84	25 51	74.49	7 950	10 0	18.0	187	12 4	23.0
26/Jun/95	Cottage	2	B	9 1	34.3	1 8	43 88	10 46	33 42	23 84	76 16	8.183	9 0	22 0	87	12.4	23 0
26/Jun/95	Bens Gutter	3	A	8 8	34.1	1.9	7 62	2 78	4.84	36 48	63.52	1.522	7.0	12 0	31	12.4	23 0
26/Jun/95	Bens Gutter	3	B	8.8	34.1	1 9	7.20	2 72	4 48	37.78	62.22	1.713	6 0	11 0	56	12 4	23.0
26/Jun/95	Honeywood	4	A	8.8	34.1	2.7							6.0	11 0	33	12 4	23.0
26/Jun/95	Honeywood	4	B	8.8	34.1	2.7	6.86	2.80	4 06	40 82	59.18	2.569				12 4	23 0
26/Jun/95	Nemo	5	A	7.8	33.9	2.1	7.64	2.94	4.70	38 48	61.52	0 761	11.0	12 0	62	12 4	23.0
26/Jun/95	Nemo	5	B	7.8	33 9	2 1	7.74	2 94	4.80	37 98	62.02	0.952	10.0	12 0	53	12.4	23.0

Date	Site name	Site No.	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day R'fall	12 day R'fall
21/Aug/95	Marine	1	A	8.9	33 5	3 7	10 06	3.20	6.86	31.82	68.18	4.091	0 5	7 8	11.4	11.2	13.0
21/Aug/95	Marine	1	B	8.9	33.5	3.7	6 16	2 24	3.91	36.43	63.57	3.425	0.5	8.1	6.8	11.2	13 0
21/Aug/95	Cottage	2	A	9.2	33 7	3.5	8 84	3.06	5.78	34 62	65 38	6.185	2.5	10 7	20 5	11.2	13.0
21/Aug/95	Cottage	2	B	9 2	33 7	3.5	7 76	2.96	4.80	38 14	61 86	5.424	3.1	10 7	20 5	11.2	13 0
21/Aug/95	Bens Gutter	3	A	9 7	33.7	1 6	6 98	2.42	4 56	34.67	65 33	0 571	2.0	10 7	38 6	11.2	13.0
21/Aug/95	Bens Gutter	3	B	9 7	33.7	1 6										11.2	13 0
21/Aug/95	Honeywood	4	A	9 8	33.6	2 6	6.54	2.36	4 18	36 09	63 91	2 188	2.3	9 6	31 8	11.2	13 0
21/Aug/95	Honeywood	4	B	9 8	33.6	2.6	7.96	2.84	5 12	35 68	64.32	1 903	2.2	9.2	34.1	11.2	13.0
21/Aug/95	Nemo	5	A	9.8	33.6	1 6	9.16	3.02	6 14	32 97	67.03	2 379	3.1	8 1	47.7	11 2	13.0
21/Aug/95	Nemo	5	B	9.8	33.6	1.6	8 30	2 88	5.42	34.70	65.30	1.903	3.6	8 9	47.7	11 2	13 0
27/Sep/95	Marine	1	A	11.1	34 5	4 5	6 70	2 34	4.36	34 93	65 07	1.868	0 4	6 7	25 0	13 8	14 2
27/Sep/95	Marine	1	B	11.1	34 5	4.5	9 00	2 80	6.20	31.11	68 89	1.972	0.4	6 7	30 0	13 8	14 2
27/Sep/95	Cottage	2	A	12.4	34.3	4 0	9 84	2.86	6.98	29 07	70.93	1.349	1 5	7.4	40 0	13.8	14 2
27/Sep/95	Cottage	2	B	12.4	34.3	4 0	8 04	2 48	5.56	30.85	69 15	1.246	1.7	10.4	45.0	13.8	14.2
27/Sep/95	Bens Gutter	3	A	12.8	34.4	1 7	9 64	3 02	6.62	31.33	68 67	1.142	4.1	10.0	62 5	13.8	14.2
27/Sep/95	Bens Gutter	3	B	12 8	34.4	1 7	8 86	2.58	6 28	29 12	70.88	1.142	2 6	8 9	82 5	13.8	14.2
27/Sep/95	Honeywood	4	A	12.8	34.4	2 5	9.52	2.76	6.76	28 99	71 01	1.349	2.6	8 1	47 5	13 8	14.2
27/Sep/95	Honeywood	4	B	12 8	34.4	2 5	8.28	2.68	5 60	32 37	67.63	0.934	1 3	7 4	45 0	13 8	14.2
27/Sep/95	Nemo	5	A	11 7	34.5	2 0	14.90	4.10	10 80	27 52	72 48	2 491	3 3	8 9	57.5	13 8	14.2
27/Sep/95	Nemo	5	B	11.7	34.5	2 0	12.44	3 42	9 02	27.49	72 51	1 765	2 0	8 1	57.5	13 8	14.2
25/Oct/95	Marine	1	A	13 0	34.2	3 0	7.92	2.54	5 38	32.07	67.93	1 038	0.6	7 3	19.0	6 4	22 6
25/Oct/95	Marine	1	B	13 0	34.2	3.0	8.04	2 54	5.50	31 59	68.41	1 453	0.4	7 7	14.3	6 4	22.6
25/Oct/95	Cottage	2	A	15.6	34.7	2 0	10.54	2 96	7 58	28.08	71.92	0 519	1.0	10 0	71.4	6.4	22 6
25/Oct/95	Cottage	2	B	15 6	34.7	2.0	11.20	3 20	8 00	28 57	71.43	0 727	0.4	10 4	76.2	6 4	22 6
25/Oct/95	Bens Gutter	3	A	15 7	34 6	1.5	23.64	4 56	19 08	19.29	80.71	1 038	0 9	12 0	81.0	6.4	22.6
25/Oct/95	Bens Gutter	3	B	15.7	34 6	1.5	61.03	10 85	50 18	17.78	82.22	3 875	0.5	13 6	100.0	6.4	22 6
25/Oct/95	Honeywood	4	A	15.8	34 6	1.5	11.76	3.24	8 52	27.55	72.45	0.934	0.8	10 0	95.2	6.4	22 6
25/Oct/95	Honeywood	4	B	15.8	34 6	1 5	12.66	3 52	9 14	27.80	72.20	0.934	0.7	10.0	95.2	6.4	22.6
25/Oct/95	Nemo	5	A	16 0	35 0	1.5	30 13	7 03	23 10	23.32	76.68	1.938	0.5	13.6	104 9	6.4	22 6
25/Oct/95	Nemo	5	B	16 0	35 0	1.5	20 55	5 10	15.45	24.82	75.18	1.278	1 4	10 4	100.0	6.4	22.6
23/Nov/95	Marine	1	A	13 5	34.7	6.0	8 96	2 96	6.00	33.04	66 96	0.934	16.2	12.5	44.4	13 2	18.6
23/Nov/95	Marine	1	B	13 5	34.7	6.0	6 22	2 20	4.02	35 37	64 63	0.727	14.7	10.8	44.4	13 2	18.6
23/Nov/95	Cottage	2	A	16.3	34.2	2 2	11 22	3.04	8.18	27 09	72 91	1.142	1.6	9 6	72.2	13.2	18 6
23/Nov/95	Cottage	2	B	16.3	34 2	2.2	9 66	3.44	6 22	35.61	64.39	1 038	1.1	9 6	74.1	13 2	18 6
23/Nov/95	Bens Gutter	3	A	17.0	34.5	1 5	12 74	3.78	8 96	29 67	70.33	1.246	0.8	8 8	81.5	13.2	18 6
23/Nov/95	Bens Gutter	3	B	17 0	34 5	1.5	41 00	9.32	31 68	22 73	77.27	2 768	3.6	14 6	94.4	13 2	18.6
23/Nov/95	Honeywood	4	A	16.2	34.2	2.0	10 12	3.12	7 00	30 83	69.17	0.934	1.4	10 4	79.6	13.2	18 6
23/Nov/95	Honeywood	4	B	16.2	34.2	2.0	9.80	3.20	6.60	32.65	67.35	1.038	1 4	9 6	90.7	13.2	18 6
23/Nov/95	Nemo	5	A	16 2	34 4	1.5	10 90	3.64	7.26	33.39	66 61	1.246	0 3	8.1	115.7	13 2	18.6
23/Nov/95	Nemo	5	B	16 2	34.4	1 5	11.28	3.48	7 80	30.85	69 15	1.038	0.3	8.1	96.3	13 2	18.6
18/Dec/95	Marine	1	A	15.5	35 5	3 0										9 0	9.8
18/Dec/95	Marine	1	B	15.5	35 5	3.0	17.64	8.70	8 94	49.32	50.68	0.623	1.1	8.3	32.0	9.0	9.8
18/Dec/95	Cottage	2	A	15.9	34 6	2 5	8.52	3 60	4.92	42.25	57.75	0.415	1.8	9 2	52.0	9.0	9 8

Date	Site name	Site No.	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day R'fall	12 day R'fall
18/Dec/95	Cottage	2	B	15.9	34 6	2.5	8.94	3.60	5 34	40.27	59.73	0 208	1.5	10 0	56.0	9.0	9.8
18/Dec/95	Bens Gutter	3	A	16.1	34 8	1.0	17 20	5.30	11 90	30.81	69.19	0.727	2 7	10.7	80.0	9.0	9.8
18/Dec/95	Bens Gutter	3	B	16 1	34.8	1 0	16 40	4 74	11 66	28 90	71.10	0.519	1.6	11.5	72.0	9.0	9.8
18/Dec/95	Honeywood	4	A	16.0	34 4	2.0	10 00	3.68	6 32	36 80	63.20	0.727	1.8	9.2	48.0	9.0	9.8
18/Dec/95	Honeywood	4	B	16.0	34 4	2.0	9 92	3.78	6.14	38.10	61.90	0 623	1.9	9 2	44.0	9.0	9.8
18/Dec/95	Nemo	5	A	16 1	35.0	1 2	12.18	4.02	8.16	33.00	67.00	1.038	2.9	9 6	56.0	9.0	9 8
18/Dec/95	Nemo	5	B	16.1	35 0	1 2	10.88	4 04	6 84	37.13	62.87	1 038	1.6	9 2	56.0	9 0	9 8
25/Jan/96	Marine	1	A													48 0	48 0
25/Jan/96	Marine	1	B													48.0	48 0
25/Jan/96	Cottage	2	A	19 1	33 5	2.5	7 66	2 24	5 42	29.24	70 76	0.623	2.1	8 0	58.3	48.0	48 0
25/Jan/96	Cottage	2	B	19.1	33 5	2.5	8 20	2.34	5.86	28 54	71 46	0 623	1.8	8 0	58.3	48.0	48.0
25/Jan/96	Bens Gutter	3	A	19 8	32.9	2 2	8.12	2 48	5 64	30.54	69.46	0.727	1 6	6 4	70.8	48.0	48 0
25/Jan/96	Bens Gutter	3	B	19.8	32 9	2.2	7 74	2.18	5 56	28 17	71 83	0.830	1 6	6.4	70.8	48 0	48 0
25/Jan/96	Honeywood	4	A	18 5	32 8	2 5	9 50	2 82	6.68	29 68	70 32	0.934	2.1	7.2	52.1	48 0	48 0
25/Jan/96	Honeywood	4	B	18 5	32 8	2 5	10 02	2.84	7.18	28 34	71.66	1.038	3.9	7.6	47.9	48.0	48.0
25/Jan/96	Nemo	5	A	18 4	32 8	1.4	11 90	4.02	7.88	33 78	66.22	1.557	2.0	6.0	66.7	48 0	48.0
25/Jan/96	Nemo	5	B	18 4	32.8	1.4	10 38	3.54	6.84	34.10	65.90	1.557	1 6	6.0	54.2	48 0	48.0
14/Feb/96	Marine	1	A	15.4	34.0	5.5	10.06	3.90	6 16	38 77	61.23	1.246	1.0	9 6	122.9	70 4	77 2
14/Feb/96	Marine	1	B	15.4	34 0	5 5	7.70	3 24	4 46	42 08	57.92	0 934	0 2	9.2	208 3	70 4	77 2
14/Feb/96	Cottage	2	A	16.7	33.6	2 3	9.96	3.92	6 04	39 36	60.64	0.519	1.2	9 6	120.8	70.4	77.2
14/Feb/96	Cottage	2	B	16 7	33.6	2 3	9.92	3.70	6 22	37.30	62.70	0 934	1 2	9 6	104 2	70 4	77 2
14/Feb/96	Bens Gutter	3	A	17.6	33.2	1 2	9.94	3 48	6 46	35.01	64.99	0 830	1 0	8 8	83 3	70 4	77.2
14/Feb/96	Bens Gutter	3	B	17.6	33.2	1.2	10 40	3.52	6.88	33.85	66.15	0 830	1 3	8 8	116 7	70 4	77.2
14/Feb/96	Honeywood	4	A	16.6	33.9	2 2	10 66	3 78	6 88	35.46	64.54	0.727	1.3	9.2	118.8	70 4	77 2
14/Feb/96	Honeywood	4	B	16.6	33 9	2.2	10 18	3 74	6 44	36.74	63.26	0 727	1.5	9.2	120 8	70.4	77.2
14/Feb/96	Nemo	5	A	16 8	33 5	1.6	30 14	7.34	22 80	24 35	75 65	1.557	1.3	9.2	108.3	70.4	77 2
14/Feb/96	Nemo	5	B	16.8	33.5	1 6	13 06	4 18	8.88	32.01	67.99	0.727	2.1	9.2	102.1	70.4	77 2
12/Mar/96	Marine	1	A	16 5	34 3	4.5	13.48	4.72	8 76	35 01	64.99	1.765	1.7	7 8	150.0	0.2	9.6
12/Mar/96	Marine	1	B	16.5	34 3	4.5	8 24	3.50	4 74	42 48	57 52	1.765	1.7	7 8	41.3	0.2	9.6
12/Mar/96	Cottage	2	A	16 2	34 5	2.6	8 76	3.22	5.54	36 76	63.24	0 830	1 9	9 8	63 5	0 2	9 6
12/Mar/96	Cottage	2	B	16 2	34 5	2 6	7.14	2.70	4 44	37.82	62 18	0 830	4 3	10 9	67 6	0 2	9.6
12/Mar/96	Bens Gutter	3	A	15.8	34 4	1.2	13.20	4.00	9 20	30 30	69.70	1.246	1.0	9 2	50.4	0 2	9 6
12/Mar/96	Bens Gutter	3	B	15 8	34.4	1 2	10 06	3 22	6.84	32.01	67.99	0 311	1 4	7.4	152 1	0.2	9.6
12/Mar/96	Honeywood	4	A	17.2	35 4	1.9	10 40	3.60	6 80	34 62	65.38	0.830	1.7	8.2	73.7	0.2	9.6
12/Mar/96	Honeywood	4	B	17 2	35.4	1 9	9.92	3.42	6 50	34.48	65.52	1.142	1.6	9.0	77.7	0.2	9.6
12/Mar/96	Nemo	5	A	17.4	35.2	1.3	8 84	3.12	5.72	35 29	64 71	0.830	0.8	8.2	77.7	0.2	9.6
12/Mar/96	Nemo	5	B	17.4	35.2	1 3	9 08	3 46	5.62	38.11	61.89	0 934	0.9	8.6	76.8	0.2	9.6

Appendix 1.3.3. Little Swanport

Date	Site name	Site No	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
11/Mar/95	Marine	1	A	18.0	34.9	5 0	4 24	1.84	2 40	43.36	56.64	0 643	1.3	8 1	52	7 6	7 6
11/Mar/95	Marine	1	B	18.0	34.9	5.0	4.84	1.74	3.10	35 92	64.08	0.368	0.8	8 5	57	7.6	7.6
11/Mar/95	Shack	3	A	20.7	34.9	2 0	5 03	2 05	2.98	40 73	59 27	0 459	1 0	7.7	74	7.6	7.6
11/Mar/95	Shack	3	B	20.7	34.9	2 0	6 18	2 47	3 72	39 89	60.11	0 368	1.0	7.3	77	7.6	7.6
11/Mar/95	Jacks Is	4	A	19.2	34.9	2 4	7 42	2 52	4 90	33.93	66.07	1 378	1.0	8.1	129	7 6	7.6
11/Mar/95	Jacks Is	4	B	19.2	34.9	2.4	7.22	2 42	4 80	33 49	66 51	1 378	0 6	8 5	250	7 6	7.6
11/Mar/95	Plentiful Pt	5	A	19 4	34 9	2.4	9.43	3 45	5 98	36.57	63 43	1.470	0 6	7 3	102	7.6	7 6
11/Mar/95	Plentiful Pt	5	B	19.4	34 9	2 4	8 95	3.15	5 80	35 20	64 80	1.562	0.6	7 7	139	7.6	7.6
11/Mar/95	Dyke	6	A	19 6	34 8	1 8	14.87	4 53	10 33	30.49	69 51	1.838	0.3	6.9	171	7 6	7.6
11/Mar/95	Dyke	6	B	19.6	34.8	1 8	9 27	3 42	5 85	36.87	63.13	2 205	0.6	6 5	202	7 6	7 6
27/Apr/95	Marine	1	A	14.4	35.6	9 0	10 07	2 81	7 26	27.94	72.06	1 011	8.7	11.9	90	0 0	8 0
27/Apr/95	Marine	1	B	14.4	35.6	9 0	4 97	1 67	3.30	33 62	66.38	0.735	8 3	10 4	66	0 0	8 0
27/Apr/95	Limekiln	2	A	13 0	35 5	4.0	5.90	2 22	3.68	37.63	62 37	0 459	1 6	8.1	73	0.0	8 0
27/Apr/95	Limekiln	2	B	13.0	35 5	4.0	5.80	2 18	3 62	37 59	62.41	0.551	1 1	9 2	73	0.0	8.0
27/Apr/95	Shack	3	A	13 4	35.5	2 0	5 60	2 23	3 37	39 88	60.12	0 643	0.8	8.1	96	0 0	8.0
27/Apr/95	Shack	3	B	13 4	35 5	2.0	5 63	2.20	3.43	39 05	60 95	0 735	0.7	7.7	75	0 0	8 0
27/Apr/95	Jacks Is	4	A	12.9	35 5	2 4	6 74	2 38	4 36	35.31	64 69	0 643	0.8	7 3	146	0.0	8.0
27/Apr/95	Jacks Is	4	B	12.9	35.5	2 4	6 62	2 18	4 44	32 93	67.07	1 011	1.9	8 8	84	0.0	8 0
27/Apr/95	Plentiful Pt.	5	A	12 2	35.1	2.6	6 06	2 34	3.72	38 61	61.39	1.011	2 5	5.8	202	0.0	8 0
27/Apr/95	Plentiful Pt	5	B	12 2	35 1	2.6	7 24	2 68	4.56	37 02	62 98	1.011	1.8	6.5	198	0 0	8.0
27/Apr/95	Dyke	6	A	11 9	34 7	2 6	8.16	2 88	5 28	35.29	64 71	1.286	2.8	5 0	250	0 0	8.0
27/Apr/95	Dyke	6	B	11.9	34 7	2 6	7 56	2.64	4 92	34.92	65 08	1.194	2 3	5.0	252	0.0	8 0
25/May/95	Marine	1	A	13 5	35 6	8 0	5.12	2.22	2.90	43.36	56 64		7 0	11.2	88	0 0	0 0
25/May/95	Marine	1	B	13.5	35 6	8.0	12.62	3 88	8.74	30.74	69.26	0 092	10 5	11 9	57	0.0	0 0
25/May/95	Limekiln	2	A	12.2	35 8	3 8	5.78	2 38	3.40	41.18	58 82	0.919	2.0	9.2	67	0 0	0.0
25/May/95	Limekiln	2	B	12.2	35.8	3.8	5.66	2 32	3 34	40.99	59 01	0.827	2.0	9.6	68	0 0	0 0
25/May/95	Shack	3	A	12.0	35.7	2 0	6.42	2 50	3 92	38.94	61 06	2.113	1 8	9 6	70	0 0	0 0
25/May/95	Shack	3	B	12 0	35.7	2 0	7.06	2.64	4.42	37.39	62 61	0.490	1 5	9.2	138	0 0	0 0
25/May/95	Jacks Is	4	A	12.2	35 7	2.4						0.000				0.0	0.0
25/May/95	Jacks Is	4	B	12.2	35 7	2 4	5.80	2 32	3 48	40.00	60 00	1.103	2.0	8.1	45	0 0	0 0
25/May/95	Plentiful Pt.	5	A	11.5	35 6	3.6	6 40	2 66	3.74	41.56	58.44	1 286	1 5	7.3	102	0 0	0.0
25/May/95	Plentiful Pt.	5	B	11.5	35.6	3.6	6.48	2 66	3.82	41.05	58 95	1.378	3 9	7.3	105	0 0	0 0
25/May/95	Dyke	6	A	9.7	35.3	2 5	7.12	2.76	4 36	38.76	61.24	0.919	3 3	3.8	208	0.0	0.0
25/May/95	Dyke	6	B	9.7	35.3	2 5	8 08	2.86	5 22	35 40	64.60	1.838	4.5	3 7	212	0 0	0.0
27/Jun/95	Marine	1	A	10 8	35.2	8.5	11 66	3.58	8 08	30 70	69.30	0 919	54.0	18.0	40	1.2	4.4
27/Jun/95	Marine	1	B	10.8	35 2	8.5	4 62	1.88	2.74	40 69	59.31	0 459	55.0	18.0	82	1.2	4 4
27/Jun/95	Limekiln	2	A	10 0	35.2	4.5	5.58	2 22	3.36	39.78	60.22	0 551	29 0	14 0	49	1 2	4.4
27/Jun/95	Limekiln	2	B	10.0	35.2	4 5	5 04	2.08	2 96	41 27	58.73	0.459	29.0	15.0	62	1.2	4.4

Date	Site name	Site No	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
27/Jun/95	Shack	3	A	9.5	35 1	2 0	5.36	1.74	3 62	32.46	67 54	0 368	18.0	14 0	33	1.2	4 4
27/Jun/95	Shack	3	B	9.5	35 1	2 0	5.94	2.10	3.84	35 35	64 65	0.368	18 0	13.0	56	1.2	4 4
27/Jun/95	Jacks Is	4	A	10 0	35.2	2.4	5 66	2.10	3.56	37.10	62 90	0 551	29.0	14 0	38	1.2	4 4
27/Jun/95	Jacks Is	4	B	10 0	35.2	2.4	5.64	2.06	3.58	36.52	63.48	0 184	29.0	15 0	36	1.2	4 4
27/Jun/95	Plentiful Pt.	5	A	8.7	35.3	3.7	7.06	2 36	4.70	33 43	66.57	1.103	15 0	11.0	60	1.2	4.4
27/Jun/95	Plentiful Pt	5	B	8.7	35 3	3.7	7.68	2.28	5 40	29.69	70.31	0.735	15.0	10 0	69	1 2	4 4
27/Jun/95	Dyke	6	A	7.7	35 2	3.1	7.26	2 30	4 96	31.68	68 32	0.919	5 0	6 0	93	1 2	4 4
27/Jun/95	Dyke	6	B	7 7	35.2	3.1										1.2	4.4
23/Aug/95	Marine	1	A	10 9	35 2	10 0	7.64	2.50	5 14	32.73	67 27	2.105	29 1	13.3	56 8	9 0	9 0
23/Aug/95	Marine	1	B	10 9	35 2	10 0	4.51	1 61	2 90	35 73	64 27	1.905	29 3	13 3	36.4	9 0	9 0
23/Aug/95	Limekiln	2	A	11 5	34 0	3 5	5 57	2.07	3 50	37 13	62 87	2 406	3 8	7.4	65 9	9.0	9 0
23/Aug/95	Limekiln	2	B	11.5	34 0	3 5	5 42	1.77	3 65	32 62	67 38	2 305	3.8	7.4	65.9	9.0	9 0
23/Aug/95	Shack	3	A	11 5	33 6	2 1	5 68	1.84	3.84	32 39	67 61	2.807	0 8	5.4	75 0	9 0	9 0
23/Aug/95	Shack	3	B	11.5	33.6	2 1										9 0	9 0
23/Aug/95	Jacks Is	4	A	11 8	33.5	2 2	6 24	2 12	4.12	33 97	66 03	2.807	2.0	7.0	56 8	9 0	9 0
23/Aug/95	Jacks Is	4	B	11 8	33.5	2.2	6.46	2 40	4.06	37 15	62.85	2 706	2.3	6 7	59 1	9 0	9.0
23/Aug/95	Plentiful Pt.	5	A	11 8	31.5	3.0	7.50	2 56	4.94	34 13	65.87	3.107	1 4	3 8	165 4	9 0	9.0
23/Aug/95	Plentiful Pt.	5	B	11.8	31.5	3.0	6.52	2 22	4.30	34.05	65 95	3.208	1 4	4 6	163 5	9 0	9 0
23/Aug/95	Dyke	6	A	11 0	32.2	2 3	9.56	3 18	6.38	33 26	66 74	6.014	0.5	5 0	171.2	9 0	9 0
23/Aug/95	Dyke	6	B	11.0	32 2	2 3	9.66	3.42	6 24	35.40	64 60	5.613	0.5	4 6	175.0	9 0	9 0
25/Sep/95	Marine	1	A	11.1	35 4	5.2	10 32	3.14	7 18	30.43	69.57	3.107	3 6	9.2	33.3	4.0	5.8
25/Sep/95	Marine	1	B	11.1	35.4	5 2	6.50	2 42	4.08	37.23	62 77	3.809	3.4	8 8	54.8	4 0	5.8
25/Sep/95	Limekiln	2	A	11.2	35 2	3 5	9 90	3 12	6 78	31.52	68 48	2.506	1 1	6.5	66 7	4 0	5.8
25/Sep/95	Limekiln	2	B	11 2	35 2	3 5	7 22	2.64	4 58	36.57	63 43	2 506	0 0	6.2	64 3	4.0	5.8
25/Sep/95	Shack	3	A	11 3	35 1	3 0	7.10	2 60	4 50	36.62	63 38	2 105	0 0	6.2	52 4	4.0	5 8
25/Sep/95	Shack	3	B	11 3	35.1	3 0										4.0	5 8
25/Sep/95	Jacks Is	4	A	11.2	35 1	2.2	7.24	2.52	4 72	34.81	65.19	2 606	0.5	6 2	59.5	4.0	5 8
25/Sep/95	Jacks Is	4	B	11.2	35 1	2 2	8.20	2 86	5.34	34 88	65 12	2 706	0.5	6 5	59.5	4 0	5 8
25/Sep/95	Plentiful Pt	5	A	11.5	34.8	3.0	7.18	2.66	4 52	37.05	62.95	1 103	0 0	4 6	122 0	4.0	5.8
25/Sep/95	Plentiful Pt.	5	B	11.5	34.8	3.0	7.56	3.06	4 50	40 48	59.52	0.702	0 0	4.2	122 0	4.0	5.8
25/Sep/95	Dyke	6	A	11.8	34.5	2.5	9 44	3.36	6 08	35.59	64.41	1.704	0 9	3.8	182.9	4.0	5.8
25/Sep/95	Dyke	6	B	11.8	34.5	2 5	8.82	3 66	5.16	41.50	58 50	1.804	0.5	4.2	187.8	4.0	5 8
23/Oct/95	Marine	1	A	12 7	35.5	7 5	6 92	2 28	4.64	32.95	67 05	0.200	0.8	7.6	46.3	12 0	22 0
23/Oct/95	Marine	1	B	12.7	35.5	7.5	6 42	2 14	4 28	33.33	66 67	0 401	0.0	6 8	124.4	12.0	22.0
23/Oct/95	Limekiln	2	A	14.5	35.4	3.0	6 42	1.86	4 56	28.97	71.03	0 601	0.4	5.8	100.0	12.0	22.0
23/Oct/95	Limekiln	2	B	14 5	35.4	3 0	8 04	2.24	5 80	27 86	72 14	0.601	0.6	6.0	112 2	12.0	22.0
23/Oct/95	Shack	3	A	14 7	35.1	2 1	6 74	1.86	4.88	27 60	72.40	0.401	0 9	6 9	73 8	12 0	22.0
23/Oct/95	Shack	3	B	14 7	35 1	2 1	6.26	1.50	4.76	23 96	76 04	0.501	0.6	6 2	124.4	12 0	22 0
23/Oct/95	Jacks Is	4	A	14.3	35 0	2.8	7 24	1 98	5.26	27.35	72.65	0 601	1.2	6 9	73 8	12.0	22.0
23/Oct/95	Jacks Is	4	B	14.3	35 0	2.8	6 98	2 08	4 90	29.80	70 20	0 702	1.2	6 5	73.8	12 0	22 0
23/Oct/95	Plentiful Pt.	5	A	14.8	34.8	3 0	9 88	2.86	7 02	28 95	71 05	0.802	0.6	4.2	112 2	12.0	22.0

Date	Site name	Site No.	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
23/Oct/95	Plentiful Pt.	5	B	14 8	34.8	3 0	7.10	2.34	4 76	32 96	67.04	1.002	0 2	4.2	114 6	12 0	22 0
23/Oct/95	Dyke	6	A	15 3	34.6	1.8	9.14	2.96	6 18	32.39	67 61	1.303	1 0	4.2	168.3	12 0	22 0
23/Oct/95	Dyke	6	B	15 3	34 6	1.8	7.70	2.40	5.30	31.17	68 83	0 902	0 6	4.2	165 9	12 0	22 0
21/Nov/95	Marine	1	A	13 5	35 1		11.38	2 80	8.58	24.60	75 40	0 501	1 6	8.1	51 9	5 6	5 6
21/Nov/95	Marine	1	B	13.5	35.1		5 82	1 80	4.02	30.93	69 07	0 501	0 6	8.1	77.8	5 6	5 6
21/Nov/95	Limekiln	2	A	14 4	33 6		6 88	1 98	4.90	28 78	71.22	0.601	0 9	6.2	121 6	5 6	5.6
21/Nov/95	Limekiln	2	B	14 4	33 6		6 50	1 98	4.52	30 46	69.54	0 702	0 9	6.5	121.6	5 6	5 6
21/Nov/95	Shack	3	A	15.1	33 5		7.28	2.34	4 94	32.14	67.86	0 501	1 1	6 2	100.0	5 6	5.6
21/Nov/95	Shack	3	B	15.1	33.5		7 72	2 10	5 62	27 20	72.80	0.601	0 9	5.8	117.6	5 6	5.6
21/Nov/95	Jacks Is	4	A	14 2	33.7		8.22	2 28	5 94	27 74	72 26	1 203	1 6	6 2	135 3	5.6	5.6
21/Nov/95	Jacks Is	4	B	14 2	33.7		8.16	2 40	5.76	29 41	70 59	1.002	1 4	6 5	137 3	5 6	5.6
21/Nov/95	Plentiful Pt	5	A	15 4	31.0		6.62	2.56	4 06	38 67	61 33	1 103	0.9	3 8	303 9	5.6	5.6
21/Nov/95	Plentiful Pt	5	B	15.4	31.0		6.22	2.24	3 98	36.01	63 99	1.103	0.8	3 8	303 9	5.6	5.6
21/Nov/95	Dyke	6	A	16 0	29.9		14.24	4.62	9 62	32.44	67 56	3.609	0.8	4 2	264 7	5.6	5.6
21/Nov/95	Dyke	6	B	16 0	29 9											5 6	5 6
30/Dec/95	Marine	1	A	15 4	35 0	4.5	9 54	5.04	4.50	52.83	47 17	0 601	0 5	5.0	81 5	15.2	155.8
30/Dec/95	Marine	1	B	15 4	35 0	4.5	8.74	3.70	5.04	42.33	57.67	0 601	0 5	5 0	63 0	15.2	155.8
30/Dec/95	Limekiln	2	A	17 0	33.5	2 5	9 52	3 74	5.78	39.29	60 71	1 303	1.5	5 8	141 7	15.2	155.8
30/Dec/95	Limekiln	2	B	17 0	33 5	2 5	8 76	3 80	4.96	43 38	56 62	1 203	1 5	6 3	133.3	15.2	155.8
30/Dec/95	Shack	3	A	17 0	32 0	1 6	8 76	3 82	4.94	43.61	56 39	1 303	0 6	5 4	202.1	15.2	155 8
30/Dec/95	Shack	3	B	17 0	32 0	1 6	7 68	3 26	4.42	42 45	57 55	1 203	1 0	4.6	181.3	15.2	155 8
30/Dec/95	Jacks Is	4	A	16 8	32 0	1 5	9 46	4.18	5 28	44 19	55 81	1.704	2 4	4 2	220.8	15 2	155 8
30/Dec/95	Jacks Is	4	B	16 8	32 0	1 5										15 2	155 8
30/Dec/95	Plentiful Pt.	5	A	17 6	30 0	2 5	8.90	3 82	5.08	42 92	57 08	2 807	2 4	3 8	460.0	15 2	155 8
30/Dec/95	Plentiful Pt.	5	B	17.6	30 0	2 5	9.22	3.66	5 56	39 70	60.30	2 907	1.1	3 3	484.0	15 2	155 8
30/Dec/95	Dyke	6	A	18.6	30.0	2 0	7.80	3.68	4 12	47 18	52 82	1 704	1.0	3 3	375 0	15 2	155.8
30/Dec/95	Dyke	6	B	18.6	30 0	2.0	9 14	4.42	4 72	48 36	51.64	1.604	1 3	3 3	366.7	15 2	155.8
23/Jan/96	Marine	1	A													2.8	7 6
23/Jan/96	Marine	1	B													2.8	7.6
23/Jan/96	Limekiln	2	A	17.3	34.6	2.4	13 22	4.06	9 16	30.71	69 29	2 005	0 4	7.2	56.3	2 8	7 6
23/Jan/96	Limekiln	2	B	17.3	34.6	2 4	12 78	4 00	8.78	31.30	68 70	2.105	2 3	8.0	45.8	2 8	7 6
23/Jan/96	Shack	3	A	17.6	34.6	2.6	8 00	2.94	5.06	36.75	63.25	1.504	0.4	8 4	52.1	2 8	7.6
23/Jan/96	Shack	3	B	17.6	34.6	2 6	35.04	9.34	25.70	26 66	73.34	3 609	2.5	10.4	31.3	2 8	7 6
23/Jan/96	Jacks Is	4	A	17 4	34.5	2 3	10.96	3 40	7 56	31.02	68 98	2 506	0.6	6 8	41.7	2.8	7 6
23/Jan/96	Jacks Is	4	B	17.4	34.5	2 3	10.64	3.40	7 24	31 95	68.05	2.205	1.0	7 2	47.9	2 8	7.6
23/Jan/96	Plentiful Pt	5	A	18.0	33.1	2.4	9 00	3 26	5.74	36 22	63.78	1.704	0 4	4 8	120.0	2.8	7 6
23/Jan/96	Plentiful Pt.	5	B	18.0	33.1	2.4	5 38	2 24	3.14	41 64	58 36	1.704	0 0	5 2	117.8	2.8	7.6
23/Jan/96	Dyke	6	A	20 6	32.1	1 4	18.58	5 30	13 28	28.53	71 47	1.704	0.2	4 8	211 1	2.8	7.6
23/Jan/96	Dyke	6	B	20.6	32 1	1.4	16.12	4.76	11.36	29.53	70 47	1.704	0 6	4.8	215.6	2 8	7 6
15/Feb/96	Marine	1	A	16.5	34 5	4 8	14.00	4 98	9 02	35.57	64 43	1 203	1 6	8 1	79 2	40 2	50 0
15/Feb/96	Marine	1	B	16.5	34.5	4.8	7.64	2.96	4 68	38 74	61.26	1 303	1.4	8.1	45 8	40 2	50.0

Date	Site name	Site No	Rep. no.	Temp (C)	Salinity (ppt)	Secchi (m)	TPM mg/l	POM mg/l	PIM mg/l	%POM	%PIM	Chl a ug/L	NOX-N ug/L	PO4-P ug/L	SiO4-Si ug/L	7 day r'fall	12 day r'fall
15/Feb/96	Limekiln	2	A	17 7	20.0	1 0	12.24	3 84	8 40	31.37	68.63	1.604	5.0	5 4	1247 0	40.2	50 0
15/Feb/96	Limekiln	2	B	17.7	20.0	1 0	13.05	4 58	8.48	35 06	64.94	1.337	4 3	6 2	1247.0	40.2	50 0
15/Feb/96	Shack	3	A	18.8	18 8	1 3	9.80	4 05	5.75	41 33	58 67	1 203	2 9	3.8	1235 0	40 2	50.0
15/Feb/96	Shack	3	B	18 8	18 8	1.3	8 90	3 43	5 47	38.48	61 52	1 069	1 8	5 0	1212 0	40 2	50 0
15/Feb/96	Jacks Is	4	A	17 2	19 7	1.1	13.45	4 65	8 80	34.57	65 43	5 079	3.9	5.8	1271 0	40 2	50 0
15/Feb/96	Jacks Is	4	B	17 2	19 7	1 1	13.28	5.28	8 00	39.74	60 26	1 203	5.5	5 4	1259.0	40 2	50.0
15/Feb/96	Plentiful Pt.	5	A	17.5	9 6	0 6	10.50	4 75	5.75	45 24	54.76	0 401	7.8	8.5	1247.0	40 2	50 0
15/Feb/96	Plentiful Pt	5	B	17.5	9 6	0 6	11.75	4 50	7 25	38 30	61.70	0.401	7.8	8 8	1235 0	40.2	50 0
15/Feb/96	Dyke	6	A	17.8	7.8	0 7	14 68	5.15	9 53	35.09	64.91	0.134	7.4	7 3	1282 0	40 2	50 0
15/Feb/96	Dyke	6	B	17.8	7.8	0 7	15 70	4 73	10 98	30 10	69.90	0 267	7 4	6 9	1282.0	40 2	50.0
13/Mar/96	Marine	1	A	16 2	35.1	4 8	7 94	2.90	5 04	36.52	63 48	3 609	1 9	10 6	25.1	0.0	5 2
13/Mar/96	Marine	1	B	16.2	35.1	4.8	7 60	2 82	4.78	37 11	62 89	3 909	1.5	10 2	25.1	0 0	5 2
13/Mar/96	Limekiln	2	A	18 6	34 0	2 6	11.36	3.38	7.98	29.75	70.25	0 601	2.8	8.6	129.2	0 0	5.2
13/Mar/96	Limekiln	2	B	18 6	34 0	2 6	7 86	2.64	5.22	33.59	66 41	1.303	2 3	8.2	127 1	0 0	5 2
13/Mar/96	Shack	3	A	19.2	33 8	1 8	5.56	1.84	3 72	33.09	66.91	0.601	2.8	9.0	81 8	0.0	5 2
13/Mar/96	Shack	3	B	19.2	33 8	1 8	5.76	2.06	3 70	35 76	64 24	0.601	2 0	8 8	93 9	0 0	5 2
13/Mar/96	Jacks Is	4	A	18.6	32.9	2 8	6.52	2 24	4.28	34 36	65 64	1 002	2 2	7 8	133.3	0 0	5.2
13/Mar/96	Jacks Is	4	B	18.6	32.9	2 8	7 16	2 38	4 78	33 24	66 76	1 002	2 2	7 8	133.3	0 0	5.2
13/Mar/96	Plentiful Pt	5	A	19.1	32.8	3.2										0 0	5.2
13/Mar/96	Plentiful Pt.	5	B	19.1	32 8	3.2	7 54	2.62	4 92	34.75	65 25	1 704	0.8	6.2	229.2	0 0	5.2
13/Mar/96	Dyke	6	A	18 6	32 3	2 5	8 20	2 72	5 48	33.17	66 83	1 504	1.2	6 2	375.0	0.0	5.2
13/Mar/96	Dyke	6	B	18 6	32.3	2 5	7.80	2.64	5 16	33.85	66.15	1 002	1 7	6 2	397 9	0.0	5.2

Appendix 2. ANOVA tables - biodeposition study

Table 2.1. ANOVA comparison of Summer TPM (g dw basket^{-1} day^{-1}) between oyster and control traps.

Source	df	MS	F	P
Trap type	1	2410.18	62.97	< 0.005
Residual	33(1)	38.27		

Table 2.2. ANOVA comparison of Winter TPM (g dw basket^{-1} day^{-1}) between oyster and control traps.

Source	df	MS	F	P
Trap type	1	77.462	70.16	0.001
Residual	20(2)	1.104		

Table 2.3. ANOVA comparison of TPM (g dw basket^{-1} day^{-1}) for two the periods (summer & winter).

Source	df	MS	F	P
Sampling period	1	1274.41	19.30	< 0.001
Residual	55(3)	66.03		

Table 2.4. ANOVA results of daily TPM (g dw basket^{-1} day^{-1}) values for each of the sampling days in summer.

Source	df	MS	F	P
Daily	2	254.26	2.68	ns
Residual	32(1)	94.99		

Table 2.5. ANOVA results of daily TPM (g dw basket^{-1} day^{-1}) values for each of the sampling days in winter.

Source	df	MS	F	P
Daily	1	0.842	0.18	ns
Residual	20(2)	4.763		

Table 2.6. ANOVA of %POM values for oyster and control traps in summer.

Source	df	MS	F	P
Trap type	1	1.012	0.29	ns
Residual	33(1)	3.550		

Table 2.7. ANOVA of %POM values for oyster and control traps in winter.

Source	df	MS	F	P
Trap type	1	125.481	14.28	0.005
Residual	21(1)	8.790		

Table 2.8. ANOVA of %POM content of sediments under oyster baskets and at control sites in summer.

Source	df	MS	F	P
Trap type	1	0.0415	0.19	ns
Residual	8(2)	0.2224		

Table 2.9. ANOVA of %POM content of sediments under oyster baskets and at control sites in winter.

Source	df	MS	F	P
Trap type	1	6.955	0.70	ns
Residual	10	9.891		

Table 2.10. Mean (± sd) total particulate matter (mg L) of tidal flux study conducted over two days in Pipeclay Lagoon.

Date: 15/6/96 (Saturday)

Sample time	Mean TPM	SD TPM
9:30 AM	9.11	0.65
11:00 AM	7.20	0.65
12:30 PM	7.26	0.35
2:00 PM	7.58	0.31
3:30 PM	7.49	0.36

Date: 16/6/96 (Sunday)

Sample time	Mean TPM	SD TPM
10:20 AM	7.29	1.44
11:30 AM	7.20	0.52
1:00 PM	7.03	0.62
2:30 PM	6.94	0.34
4:35 PM	7.08	0.43

Appendix 3 Summary tables - oyster growth and condition.

Table 3.1 Mean (± sd) shell length, live weight, dry shell weight, dry meat weight and shell cavity capacity of oysters at two sites in Pitt Water. Initial values and those at 3 and 6 months. Percentage increase determined from comparison with initial values. (n = 40)

Trial 1		Site 1				Site 2			
	Initial	3 months	% increase	6 months	% increase	3 months	% increase	6 months	% increase
Length (mm)	65 ± 6	72 ± 6	10	74 ± 7	14	64 ± 7	-2	64 ± 5	-1
Weight (g)	28 4 ± 4.8	38 5 ± 6.5	35.6	43.1 ± 7.2	51 8	35.5 ± 4 9	25.2	37.9 ± 7.8	33 5
Dry shell wt. (g)	15.67 ± 3.23	21 98 ± 4 51	40.3	26 88 ± 7.72	71.5	21 57 ± 3 45	37.6	24.51 ± 4 48	56 4
Dry meat wt. (g)	0 62 ± 0 19	1.11 ± 0 21	80.1	1.21 ± 0 22	95.7	1.04 ± 0.17	68 5	1.16 ± 0.19	87.0
Shell cavity capacity	12.7 ± 2.10	15.52 ± 6 14	29.9	13 04 ± 2.99	2 54	13.96 ± 1.90	9 8	10.97 ± 1 85	-13 8

Table 3.2 Summary table of initial and final condition indices data for Trials 2 and 3, two sites in Pitt Water (n = 40)

		Site 1		Site 2	
Trial 2	Initial	Final	% Increase	Final	% Increase
Length (mm)	68 ± 5	81 ± 5	19	76 ± 6	12
Weight (g)	26 2 ± 4.4	52.3 ± 9.7	99.3	48.9 ± 6 5	86.4
Dry shell wt. (g)	13.15 ± 2 30	29 87 ± 5.32	127.1	28 16 ± 3 90	114.1
Dry meat wt. (g)	0.54 ± 0 11	1 75 ± 0.40	221.7	1.55 ± 0 21	184.0
Shell cavity capacity	11 08 ± 2.70	22 41 ± 5 32	102 3	20 73 ± 3 69	87.1
Trial 3					
Length (mm)	78 ± 7	87 ± 8	11	92 ± 7	17
Weight (g)	44 8 ± 8.2	58 7 ± 10.9	30 8	63.3 ± 9.2	41.3
Dry shell wt. (g)	23.00 ± 4 40	37.51 ± 7 44	63.1	37.42 ± 6.02	62 7
Dry meat wt (g)	1.52 ± 0.36	1 88 ± 0 40	24 2	1 78 ± 0 42	17.1
Shell cavity capacity	21.83 ± 4.39	21.15 ± 4.62	-3.1	25.9 ± 4 02	18 7

Table 3.3 Summary of mean (± sd) Crosby & Gale and Lucas & Beninger Condition index values for initial and final data for trials 1, 2 and 3 at two sites in Pitt Water (n = 40/site)

		Site 1		Site 2	
Trial 1	Initial	Final	% Increase	Final	% Increase
Crosby & Gale	48.57 ± 13 24	75 53 ± 10 19	55.5	86.92 ± 10.83	79 0
Lucas & Beninger	0 0392 ± 0 0091	0 0456 ± 0.0076	16.16	0 0478 ± 0.0071	21 9
Trial 2					
Crosby & Gale	42.59 ± 8 17	79 96 ± 15.85	87 7	76 34 ± 14.62	79.2
Lucas & Beninger	0 0414 ± 0.0053	0 0588 ± 0 0094	41.9	0 0554 ± 0 0082	33 8
Trial 3					
Crosby & Gale	69 62 ± 9 02	90 51 ± 15.81	30 0	68 38 ± 10.63	-1.8
Lucas & Beninger	0 0661 ± 0.0089	0 0505 ± 0.0065	-23 5	0 0475 ± 0.0084	-28.1

Table 3.4 Summary table of nested two-way ANOVA of final shell length, live weight, dry shell weight, dry meat weight, shell cavity capacity, Crosby & Gale condition index, and Lucas & Beninger condition index for two sites, over three trials at Pitt Water. (n=40). NB. Only results of a one-way ANOVA are shown for final results of trial 2 (refer to text).

Trial	Variable	Site	Final Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)
1	Length (mm)	1	74.35a	0.005	75.40	79.10	72 60	70 30	ns
		2	64.47b		63.70	63.90	66.00	64.30	
	Weight (g)	1	43 09a	0.050	45.08	45.04	43.30	38.92	ns
		2	37.90b		36.86	39.04	38.78	36.91	
	Dry shell weight (g)	1	26.88	ns	27.55	27 25	27.85	24.88	ns
		2	24.51		24 17	25 87	24.87	23.12	
	Dry meat weight (g)	1	1.21	ns	1 23	1.29	1.19	1.13	ns
		2	1.16		1 07	1 17	1.24	1.15	
	Shell cavity capacity	1	13.04	ns	14 08ab	14.71a	12.2bc	11.17c	0.050
		2	10.97		10.45a	10.61a	11.35a	11.45a	
	Crosby & Gale CI	1	75.5b	0.010	70.90	73.70	77.50	80.00	ns
		2	86.9a		85.30	89.20	89.50	83.60	
	Lucas & Beninger CI	1	0.0456	ns	0.0450	0.0482	0.0431	0 0460	ns
		2	0 0478		0.0452	0.0453	0 0506	0.0502	
2	Length (mm)	1	81 25a	<0.001					
		2	76.45b						
	Weight (g)	1	52.28	ns					
		2	48 89						
	Dry shell weight (g)	1	29.87	ns					
		2	28 16						
	Dry meat weight (g)	1	1.75a	0.020					
		2	1.55b						
	Shell cavity capacity	1	22.41	ns					
		2	20.73						
	Crosby & Gale CI	1	80.00	ns					
		2	76.30						
	Lucas & Beninger CI	1	0.0589	ns					
		2	0.0555						
3	Length (mm)	1	87.07b	0.050	89.20	85.90	87.50	85.70	ns
		2	91.62a		89.90	94.80	91.80	90.00	
	Weight (g)	1	58.70	ns	62.20	58.90	58.80	54.80	ns
		2	63.30		59.70	69.00	62.90	61.70	
	Dry shell weight (g)	1	37.51	ns	39.43	36.88	38.15	35.56	ns
		2	37.42		34.39	41.38	37.49	36 42	
	Dry meat weight (g)	1	1.89	ns	1 96	1.84	1.83	1.92	ns
		2	1.77		1.68	1.87	1.81	1.75	
	Shell cavity capacity	1	21.15b	0 010	22.78	21.99	20.63	19.20	ns
		2	25.90a		25.34	27.57	25.45	25.26	
	Crosby & Gale CI	1	90.50a	<0 001	88.90	84.90	88.50	99.70	ns
		2	68.40b		65.10	68.00	70.40	69.90	
	Lucas & Beninger CI	1	0.0505	ns	0 0497	0.0498	0.0483	0.0541	ns
		2	0.0475		0.0484	0.0449	0.0482	0.0486	

Table 3.5 Summary table of mean (± sd) condition indices data for trials 1, 2 and 3 at two sites in Pipeclay Lagoon (n = 40/site)

		Site 1 (South)		Site 2 (North)	
Trial 1	Initial	Final	% Increase	Final	% Increase
Length (mm)	69 ± 6	85 ± 8	23	84 ± 6	21
Weight (g)	28 8 ± 4.1	46 3 ± 7.0	61	47.9 ± 9.0	66.5
Dry shell wt. (g)	14 06 ± 2 27	25 74 ± 4 41	83.1	28 08 ± 5 56	99 8
Dry meat wt. (g)	0 62 ± 0 17	1 15 ± 0 23	83 2	1 34 ± 0 34	114.7
Shell cavity capacity	14 69 ± 2 25	20 53 ± 3 58	39 8	19.78 ± 4.14	34 6
Trial 2					
Length (mm)	68 ± 8	83 ± 8	22	74 ± 9	8
Weight (g)	29 5 ± 8.6	53.3 ± 12.9	80.8	41.1 ± 9.2	39.2
Dry shell wt. (g)	16 96 ± 5.25	32.7 ± 8.57	92.8	25.80 ± 6.19	52.1
Dry meat wt. (g)	0 87 ± 0 31	1.76 ± 0.50	103.4	1.23 ± 0.33	42.0
Shell cavity capacity	12.55 ± 3.63	20 64 ± 5 07	64.5	15 27 ± 3.36	21.7
Trial 3					
Length (mm)	66 ± 5	85 ± 9	29	85 ± 8	28
Weight (g)	37 9 ± 5 8	57 8 ± 10 7	52.6	62.2 ± 7 4	64 3
Dry shell wt. (g)	23.66 ± 3 92	37.55 ± 7.33	58.7	41.00 ± 4 87	73.2
Dry meat wt. (g)	1 18 ± 0.28	1.40 ± 0.31	18.8	1 70 ± 0.34	44.2
Shell cavity capacity	14.19 ± 2.29	20 23 ± 4 17	42.6	21.20 ± 3.57	49 4

Table 3.6 Summary of mean (± sd) Crosby & Gale and Lucas & Beninger Condition index values for initial and final data for trials 1, 2 and 3 at two sites in Pipeclay Lagoon (n = 40/site)

		Site 1	(South)	Site 2	(North)
Trial 1	Initial	Final	% Increase	Final	% Increase
Crosby & Gale	42 75 ± 10 50	56 47 ± 10 16	32.1	68.18 ± 11.58	59.5
Lucas & Beninger	0.0447 ± 0.0103	0.0448 ± 0.0067	0.2	0 0478 ± 0 0077	6 9
Trial 2					
Crosby & Gale	50.98 ± 8.01	85.42 ± 13.81	67.6	80 99 ± 14 39	58.9
Lucas & Beninger	0 0510 ± 0.0080	0.0543 ± 0.0093	6 4	0.0480 ± 0 0076	-5.9
Trial 3					
Crosby & Gale	83 26 ± 15 18	71.02 ± 19.40	-14 7	81 08 ± 15.55	-2 6
Lucas & Beninger	0 0506 ± 0.0135	0 0377 ± 0.0075	-25 5	0 0416 ± 0 0075	-17.8

Table 3.7 Summary table of nested two-way ANOVA of final shell length, live weight, dry shell weight, dry meat weight, shell cavity capacity, Crosby & Gale condition index, and Lucas & Beninger condition index for two sites, over three trials at Pipeclay Lagoon. (n=40).

Trial	Variable	Site	Final Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)
1	Length (mm)	1	85.31	ns	83 40	91 32	85 00	81.50	ns
		2	83.95		83 20	82.80	84.60	85.20	
	Weight (g)	1	46.27	ns	46.36	49.70	43.67	45.36	ns
		2	47.86		47.16	46.01	50.00	48.28	
	Dry shell weight (g)	1	25.74	ns	25.75	27.80	24.87	24.53	ns
		2	28.08		27.93	26.33	29.79	28.26	
	Dry meat weight (g)	1	1.15b	0.050	1.21	1.20	1.10	1.08	ns
		2	1.34a		1.36	1.28	1.48	1.24	
	Shell cavity capacity	1	20.53	ns	20.61	21.90	18.80	20.83	ns
		2	19.78		19.23	19.67	20 21	20.02	
	Crosby & Gale CI	1	56.5b	0.050	59.20	55.60	58.60	52.50	ns
		2	68 2a		70.90	64.80	74.00	63.00	
	Lucas & Beninger CI	1	0.0448	ns	0.0471	0.0432	0.0445	0.0443	ns
		2	0.0478		0.0490	0.0480	0.0494	0.0446	
2	Length (mm)	1	83.45a	<0.001	85 00	80.10	86.50	82.20	ns
		2	73.95b		74.80	72.10	76 50	72.40	
	Weight (g)	1	53.30a	0.010	57 80	48.70	50.80	56.00	ns
		2	41.10b		43 60	41.00	42.10	37.60	
	Dry shell weight (g)	1	32.70a	0.050	36.41	29.47	30 12	34.81	ns
		2	25.80b		27.36	25.91	26.10	23.84	
	Dry meat weight (g)	1	1 76a	0.050	2.05	1 51	1.62	1.86	ns
		2	1.23b		1.26	1.29	1.22	1.16	
	Shell cavity capacity	1	20.64a	<0.001	21.38	19 23	20.71	21.24	ns
		2	15.27b		16.24	15.09	16.01	13.74	
	Crosby & Gale CI	1	85.40	ns	97.1a	78.4b	78.4b	87.8ab	0.050
		2	81.00		77.4a	84.7a	76.7a	85.1a	
	Lucas & Beninger CI	1	0.0543a	0.050	0.0580	0.0510	0 0530	0.0550	ns
		2	0.0480b		0.0467	0.0492	0.0466	0.0493	
3	Length (mm)	1	85.12	ns	84.20	90 00	81.50	84.80	ns
		2	84.80		85.20	87.30	84.60	82.10	
	Weight (g)	1	57.80	ns	59.00	62.10	51.30	58.60	ns
		2	62.20		62.70	65.00	60.90	60.30	
	Dry shell weight (g)	1	37.55	ns	37.64	40 59	33.64	38.33	ns
		2	41.00		40.10	43.36	40 66	39.86	
	Dry meat weight (g)	1	1.40b	0.050	1.36	1 51	1.33	1.40	ns
		2	1.70a		1.67	1 82	1.76	1.55	
	Shell cavity capacity	1	20.23	ns	21.37	21.54	17.70	20.32	ns
		2	21.20		22.56	21.60	20.23	20.40	
	Crosby & Gale CI	1	71.00	ns	63.40	70.70	78.60	71.50	ns
		2	81.10		75.30	85.50	88.00	75.40	
	Lucas & Beninger CI	1	0.0377b	0.050	0.0362	0 0378	0.0398	0.0371	ns
		2	0.0416a		0.0417	0.0426	0 0432	0.0389	

Table 3.8 Summary table of mean (± sd) condition indices data for trials 1, 2, and 2 at two sites in Little Swanport (n = 40/site).

		Site 1	(Channel)	Site 2	(Ram Island)
Trial 1	Initial	Final	% Increase	Final	% Increase
Length (mm)	54 ± 3	86 ± 10	60	92 ± 10	70
Weight (g)	17 8 ± 2 9	62.8 ± 13.8	252 8	69 2 ± 13 7	289 2
Dry shell wt (g)	9.16 ± 1 46	36 7 ± 6.94	301 2	39.68 ± 6.90	333 3
Dry meat wt. (g)	0 69 ± 0.12	2.47 ± 0.46	259.0	2.65 ± 0 61	285.1
Shell cavity capacity	8.63 ± 1 62	26 02 ± 7 55	201 4	29 56 ± 7.30	242.4
Trial 2					
Length (mm)	63 ± 5	82 ± 7	30	91 ± 9	44
Weight (g)	19 0 ± 2 8	48 9 ± 6.2	156.7	54.9 ± 10 8	188 2
Dry shell wt. (g)	10.45 ± 1.53	28 08 ± 3 63	168 8	28 68 ± 4.96	174 6
Dry meat wt. (g)	0.76 ± 0.13	1.78 ± 0 23	132 8	1 79 ± 0 42	134 5
Shell cavity capacity	8 60 ± 1.57	20.80 ± 3 34	142.0	26.20 ± 6 31	204.8
Trial 3					
Length (mm)	53 ± 4	79 ± 6	49	77 ± 6	45
Weight (g)	15 3 ± 2 4	47 1 ± 6.5	207.3	47 6 ± 5.9	210.8
Dry shell wt (g)	8.32 ± 1 47	26 08 ± 3 54	213 4	26 96 ± 3 85	223 9
Dry meat wt. (g)	0.57 ± 0.11	1 92 ± 0.45	240.2	2 03 ± 0.35	259.9
Shell cavity capacity	6.99 ± 1 23	20 97 ± 3 41	200 0	20.63 ± 3.26	195 1

Table 3.9 Summary of mean (± sd) Crosby & Gale and Lucas & Beninger Condition index values for initial and final data for trials 1, 2 and 3 at two sites in Little Swanport (n = 40/site)

		Site 1	(Channel)	Site 2	(Ram Is.)
Trial 1	Initial	Final	% Increase	Final	% Increase
Crosby & Gale	80 64 ± 11 15	99 00 ± 19 75	22.8	90 76 ± 15 21	12 6
Lucas & Beninger	0.0754 ± 0 0080	0 0680 ± 0.0101	-9 8	0.0667 ± 0.0105	-11.5
Trial 2					
Crosby & Gale	89 78 ± 12 75	86.60 ± 12 64	-3.5	69.56 ± 12 89	-22.5
Lucas & Beninger	0 0731 ± 0 0082	0 0638 ± 0.0087	-12 8	0 0624 ± 0 0108	-14.6
Trial 3					
Crosby & Gale	81.43 ± 11 87	91 77 ± 16 45	12 7	100 08 ± 19 00	22.9
Lucas & Beninger	0 0685 ± 0 0111	0 0734 ± 0 0122	7 1	0.0760 ± 0 0127	11.0

Table 3.10 Summary table of nested two-way ANOVA of final shell length, live weight, dry shell weight, dry meat weight, shell cavity capacity, Crosby & Gale condition index, and Lucas & Beninger condition index for two sites, over three trials at Little Swanport. (n=40).

Trial	Variable	Site	Final Site (Mean)	P (site)	Basket (Mean) 1	2	3	4	P (basket/site)
1	Length (mm)	1	86.30	ns	82.70	86 30	82.70	93 60	ns
		2	91.70		86.80	90.40	93 20	96.40	
	Weight (g)	1	62.80	ns	60.30	62.70	57.40	70.50	ns
		2	69.20		66.60	66.20	71.40	72.90	
	Dry shell weight (g)	1	36.74	ns	36 18	37.36	33.26	40 14	ns
		2	39 67		38.55	38.65	41.16	40.34	
	Dry meat weight (g)	1	2.65a	0.050	2.38	2.54	2.45	2.50	ns
		2	2.47b		2.75	2.66	2.52	2.65	
	Shell cavity capacity	1	26.02	ns	24.15	25.39	24 16	30.38	ns
		2	29.56		28.01	27.50	30.16	32.56	
	Crosby & Gale CI	1	99.00	ns	100.40	102 80	104.90	88.00	ns
		2	90.80		98.90	98.40	84.00	81.70	
	Lucas & Beninger CI	1	0.0680	ns	0.0663	0.0683	0.0741	0.0632	ns
		2	0.0667		0.0717	0.0691	0.0613	0.0647	
2	Length (mm)	1	81.90b	<0.001	82.70	83.40	78.90	82.60	ns
		2	91.12a		91.10	87.50	92.90	93 00	
	Weight (g)	1	48.90b	0.050	49.90	51.90	46 90	46.90	ns
		2	54.90a		55.30	53.60	54 00	56.60	
	Dry shell weight (g)	1	28.08	ns	27.77	30.05	26.88	27.62	ns
		2	28.68		29.65	29.00	27.05	29.04	
	Dry meat weight (g)	1	1.78	ns	1.73	1.83	1.82	1.72	ns
		2	1.79		1 89	1.86	1.61	1.80	
	Shell cavity capacity	1	20.80b	<0.001	22.10	21.86	19.99	19.24	ns
		2	26.20a		25.63	24.57	26 99	27.60	
	Crosby & Gale CI	1	86.60a	0 050	78.5b	86.3ab	91 9a	89.6a	0 010
		2	69.60b		75.4a	76 6a	59.8b	66.4ab	
	Lucas & Beninger CI	1	0.0638	ns	0.0629	0 0619	0.0684	0.0619	ns
		2	0.0624		0.0641	0.0644	0.0597	0.0616	
3	Length (mm)	1	78.60	ns	80.80	76.20	78.40	79.00	ns
		2	76.55		75 50	77 70	74.00	79.00	
	Weight (g)	1	47.05	ns	47.38	44.83	50.77	45.22	ns
		2	47 59		50.18	48.38	46.57	45.22	
	Dry shell weight (g)	1	26.08	ns	26.54	25.12	28.11	24 56	ns
		2	26.96		28.34	28.01	26 93	24.56	
	Dry meat weight (g)	1	1.92	ns	2.04a	1.88a	2.07a	1 71a	0.050
		2	2.03		2.10a	2.10a	2.23a	1.71b	
	Shell cavity capacity	1	20 97	ns	20.84	19.71	22.66	20.66	ns
		2	20.63		21.83	20.37	19.64	20.66	
	Crosby & Gale CI	1	91.80	ns	99.3a	94 4a	90.7ab	82.7b	<0.001
		2	100.10		96.7b	105 3ab	115 6a	82.7c	
	Lucas & Beninger CI	1	0.0734	ns	0 0768	0.0738	0.0735	0.0694	ns
		2	0.0760		0.0756	0.0755	0.0837	0.0694	

Appendix 4 Clearance rate estimates

1. Clearance rate estimates - Smaal and Prins (1993) method

The scale of impact of bivalve culture systems and hence carrying capacity can be assessed from comparison of the residence time and clearance rate of the cultured population, that is, the time taken for the population to filter a volume equivalent to the system volume (Smaal and Prins, 1993). The model developed to calculate the values tabulated in Table 6.1 (Chapter 6) is detailed in a step-wise procedure here.

Firstly, estimation of standing stock of oysters needed to be made. As stated, actual production figures or standing stock numbers could not be obtained. Dr John Wilson determined the number of oysters of various size categories held on a lease to produce an annual production (harvest) of 1 million oysters (Table 1). These figures are provided in Crawford et al. (1996).

Table 1. Number and size category of oysters to produce 1 million per annum. (source Crawford et al., 1996).

Mean size (mm)	Number of oysters
8	742500
20	477612
50	656716
70	284119
	Total = 2160947

Thus, approximately 2.2 million oysters are held for each 1 million produced annually. However, this estimate is based on oysters attaining a market size from seed (~ 6-8 mm) to approximately 70 mm in 12 months. The average time taken to reach market size in each of the three areas is known from discussion with the oyster farmers. These were 18 months in Little Swanport, 36 months in Pipeclay Lagoon and 42 months in Pitt Water. This was therefore factored into the model by multiplying the number of oysters required to produce the per annum production figure by the average time taken to reach market size to yield estimation of the standing stock. This assumption is not unreasonable and was supported from discussions with John Thomson (Marine Biologist, oyster grower) who stated that the general rule of thumb is "for every million produced, twice this amount is held on the lease and if it takes three years to reach harvest size then there would be three times this amount held".

Standing stock (number of oysters) were calculated using the figures from Table 1 multiplied by the annual production and time taken to reach harvest. The average biomass of oysters in each size category above was calculated from the results of the oyster growth data using the average value of $DW = a L^b$, where DW = dry weight (g) and L = length (mm). Average filtration rate was taken as 50 L d^{-1} g^{-1} DW (Raillard and Ménesguen, 1994; Ball et al., 1997).

Clearance time (days) using Smaal and Prins (1993) method was determined from,

$$Clearance\ time = \frac{Mean\ volume \times 10^6 \times 10^3}{Filtration\ rate \times total\ biomass} \qquad \text{(days)}$$

where mean volume (L), filtration rate (L g^{-1} d^{-1}) and total biomass (g). The figures used in the calculations and model output for Upper Pitt Water, Pipeclay Lagoon and Little Swanport are shown.

Upper Pitt Water

Surface area of Upper Pitt Water estuary	23.6 km^2
Mean volume of Upper Pitt Water estuary	41.23 million m^3
Total area of leases	108.19 ha
Production of oysters 1995/96 (DPIF, 1998)	6.2 million oysters
Average time taken to reach harvest	3.5 years

Table 2. Estimated standing stock and total biomass of oysters held on leases in Upper Pitt Water.

Length (mm)	Standing Stock (No. Oysters)	Average biomass /oyster (g DW)	Total biomass (g DW)
8	16112250	0.0027	43460.41
20	10364180	0.0318	329342.29
50	14250737	0.3744	5334881.6
70	6165382	0.9261	5709458.4
Total:	**46892550**		**11417143**

where average biomass/oyster calculated using $DW = 1 \times 10^{-5} L^{2.6918}$

The number of oysters per m^2 total lease area = 43.34 oysters m^{-2}

The average oyster biomass = 0.243 g DW

Biomass per m^2 total surface area of Upper Pitt Water = 0.495 g m^{-2}

Biomass per m^3 mean volume of Upper Pitt Water =0.277 g m^{-3}

Clearance time for Upper Pitt Water (assuming 24 hrs feeding) was 72.22 days. Clearance time is longer if immersion period is considered, for example the clearance time of oysters exposed out of the water for 15% of the time would be 84.97 days.

Pipeclay Lagoon

Surface area of Pipeclay Lagoon	4.605 km^2
Mean volume of Pipeclay Lagoon	3.48 million m^3
Total area of leases	48.25 ha
Production of oysters 1995/96 (DPIF, 1998)	8.2 million oysters
Average time taken to reach harvest	3.0 years

Table 3. Estimated standing stock and total biomass of oysters held on leases in Pipeclay Lagoon.

Length (mm)	Standing Stock (No. Oysters)	Average biomass /oyster (g DW)	Total biomass (g DW)
8	18265500	0.0068	124858.31
20	11749255	0.0515	604696.79
50	16155214	0.3875	62601027
70	6989327	0.8132	5683991.6
Total:	**53159296**		**12673649**

where average biomass/oyster calculated using $DW = 7 \times 10^{-5} L^{2\,2032}$

The number of oysters per m^2 total lease area = 110.2 oysters m^{-2}

The average oyster biomass = 0.238 g DW

Biomass per m^2 total surface area of Pipeclay Lagoon = 2.752 g m^{-2}

Biomass per m^3 mean volume of Pipeclay Lagoon = 3.643 g m^{-3}

Clearance time for Pipeclay Lagoon (assuming 24 hrs feeding time) was 5.49 days. If immersion period is considered then, for example the clearance time of oysters exposed out of the water for 15% of the time would be 6.46 days.

Little Swanport

Surface area of Little Swanport estuary	6.321 km^2
Mean volume of Little Swanport estuary	6.55 million m^3
Total area of leases	79.5 ha
Production of oysters 1995/96 (DPIF, 1998)	4 million oysters
Average time taken to reach harvest	1.5 years

Table 4. Estimated standing stock and total biomass of oysters held on leases in Little Swanport estuary.

Length (mm)	Standing Stock (No. Oysters)	Average biomass /oyster (g DW)	Total biomass (g DW)
8	4455000	0.0087	38801.1
20	2865672	0.0730	209085.4
50	3940296	0.6112	2408394.4
70	1704714	1.3340	2274173.0
Total:	**12965682**		**4930453.9**

where average biomass/oyster calculated using $DW = 7 \times 10^{-5} L^{2\,3197}$

The number of oysters per m^2 total lease area = 16.31 oysters m^{-2}

The average oyster biomass = 0.380 g DW

Biomass per m^2 total surface area of Little Swanport estuary = 0.780 g m^{-2}

Biomass per m^3 mean volume of Little Swanport estuary = 0.752 g m^{-3}

Clearance time for Little Swanport (assuming 24 hrs feeding time) was 26.59 days. If immersion period is considered then, for example the clearance time of oysters exposed out of the water for 15% of the time would be 31.28 days.

2. Modified clearance rate estimates

Assessment was made using the filtration rate model equations developed by Kobayashi et al. (1997) for *Crassostrea gigas*. The authors used estimates based on the relationship of filtration rate as a function of dry meat weight and derived two separate equations for filtration rates of small oysters up to 2.0 g dry meat weight and large oysters greater than 2.0 g. A summary of their equations were:

$2.0 \text{ g} < W_d$; $\quad FR = 2.51\ W_d^{0.279}$ $\qquad$ Eq. 1

$2.0 \text{ g} \geq W_d$ $\quad FR = 0.117\ W_d^3 - 1.05\ W_d^2 + 3.09\ W_d + 0.133$ $\qquad$ Eq. 2

where FR is filtration rate (L filtered per oyster h^{-1}) at 20^0 C and W_d is dry meat weight (g) (Kobayashi et al., 1997).

Equation 2 was used to model filtration rates as the average dry meat weights (Tables 2, 3 and 4) were less than 2.0 g. The average biomass per oyster (Tables 2, 3 and 4) provided the W_d values. The average filtration rate for each size category of oyster (Table 1) was determined and this figure multiplied by the estimated number of oysters of that size category held (see Tables 2, 3 and 4). Population filtration rate (FR) was calculated as the sum of the total filtration rates for each oyster size category.

The modified clearance time (CT_{mod}) estimates were calculated as:

$$Clearance\ time_{mod} = \frac{Mean\ volume \times 10^6}{Population\ FR} \qquad \text{(days)}$$

where Mean volume (m^3) and Population FR (m^3 filtered by the total number of oysters held d^{-1}). The mean volumes for Upper Pitt Water, Pipeclay Lagoon and Little Swanport are those given in the preceding section.

Upper Pitt Water

The filtration rate (FR) per oyster h^{-1} and total FR, based on the estimate of the number of oysters held (Table 2) for each length (size) category are given in Table 5. The population FR is the sum of the total size category filtration rates.

Table 5. Average filtration rates of oysters held on leases in Upper Pitt Water.

Length (mm)	FR per oyster h^{-1}	Total FR of oysters in each category (L per oysters h^{-1})
8	0.141	2277098.9
20	0.230	2385153.8
50	1.149	16370592.6
70	2.187	13483473.0
	Mean FR = 0.927	
	Population FR (L h^{-1}) =	**34516318.3**

The population FR ($m^3\ d^{-1}$) = 828392 $m^3\ d^{-1}$

Clearance time $_{mod}$ = 49.77 days (assuming 24 hrs feeding).

Estimated clearance time, for example an 85% immersion period, would be 58.55 days.

Pipeclay Lagoon

The filtration rate (FR) per oyster h^{-1} and total FR, based on the estimate of the number of oysters held (Table 3) for each length (size) category are given in Table 6. The population FR is the sum of the total size category filtration rates.

Table 6. Average filtration rates of oysters held on leases in Pipeclay Lagoon.

Length (mm)	FR per oyster h^{-1}	Total FR of oysters in each category (L per oysters h^{-1})
8	0.154	2814228.2
20	0.289	3398673.5
50	1.180	19055276.5
70	2.014	14079370.9
	Mean FR = 0.909	
	Population FR (L h^{-1}) =	**39347549.1**

The population FR ($m^3\ d^{-1}$) = 944341 $m^3\ d^{-1}$

Clearance time $_{mod}$ = 3.68 days (assuming 24 hrs feeding).

Estimated clearance time, for example an 85% immersion period, would be 4.33 days.

Little Swanport

The filtration rate (FR) per oyster h^{-1} and total FR, based on the estimate of the number of oysters held (Table 4) for each length (size) category are given in Table 7. The population FR is the sum of the total size category filtration rates.

Table 7. Average filtration rates of oysters held on leases in Little Swanport.

Length (mm)	FR per oyster h^{-1}	Total FR of oysters in each category (L per oysters h^{-1})
8	0.160	712055.8
20	0.353	1011320.6
50	1.656	6525603.5
70	2.664	4541905.3
	Mean FR = 1.208	
	Population FR (L h^{-1}) =	**12790885.2**

The population FR (m^3 d^{-1}) = 306981 m^3 d^{-1}

Clearance time $_{mod}$ = 21.35 days (assuming 24 hrs feeding).

Estimated clearance time, for example an 85% immersion period, would be 25.12 days.